# COMPARISON OF NEKTON UTILIZATION OF SMOOTH CORDGRASS (SPARTINA ALTERNIFLORA) MARSH BASED ON MARSH SIZE AND DEGREE OF ISOLATION FROM LIKE HABITAT: DO SIZE AND SITE LOCATION MATTER?

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ABSTRACT

Influence of salt marsh size and proximity to like habitat on nekton use was examined. Comparisons were made among six naturally occurring isolated island marshes, three each of small (~400-1,000 $m^2$) and large (~3,000-10,000 $m^2$) island size classes and six paired mainland marshes (at least 76,000 $m^2$ in size). Two species representing opposites for dispersal ability and life history strategies, mummichog (Fundulus heteroclitus) and pinfish (Lagodon rhomboides), were used as representative species to examine population patterns. Fundulus heteroclitus exclusively utilized marsh habitats and not adjacent shallow flats during high tide while L. rhomboides utilized both. At high tide island marshes acted as a refuge for F. heteroclitus among shallow water flat habitats. Densities of marsh dependent species, including F. heteroclitus, were an order of magnitude lower within the large island compared to the mainland marshes, and nonexistent within small island marshes. Similarities among mainland and large island marshes for larvae and disparity in juvenile F. heteroclitus abundances suggested a bottleneck constrained adult and juvenile F. heteroclitus populations at large island marshes. Differential predation rates among marsh types significantly contributed to disparities in young of year abundance. Patterns of abundance suggest two crucial time periods for F. heteroclitus population maintenance, one during late spring/early summer spawn-recruitment and the second during the dispersal/emigration during the late fall/winter time period when water temperatures are colder and predator levels are significantly reduced. In contrast, L. rhomboides recruitment and dispersal occurs simultaneously during the late fall/winter when low predator levels create an optimal time for recruitment to estuaries.

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Finally, thank you to God for my life and the wonderful people in it.

# DEDICATION

This work is dedicated to those who help to impress upon me those things that make life so special:

To Eunice Ruth Meyer (1919-1999). My first mentor. Who always believed in me and taught me the importance of believing in myself.

To Lawrence Allen Meyer (1956-2002). Who showed me that the simplest things in life are not only those things to treasure the most, they are what makes life so special.

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# CHAPTER 1.

# INTRODUCTION

# INTRODUCTION

Because of the pleasing aesthetic value and proximity to navigable waters a large portion of the United States population is concentrated along the continental coastal regions, resulting in increased development along coastal rivers, creeks and sounds. This continuing trend in population growth has subsequently impacted the very estuarine habitats prized by humans and that serve as critical transition zones (CTZ's) between upland and marine habitats (Levin et al. 2001). From the 1780's through the 1980's estimated wetland losses in the contiguous U. S. approached 53% of initial totals (Dahl 1990; Johnson 1994), with comparable percentage losses of tidal salt marsh (Kennish 2001). Of the estimated 89 million acres of wetlands in the 1780's only 42 million acres were remaining by 1980 (Johnson 1994), of which ~ 5% were encompassed by the remaining emergent marsh (Zedler 1996). Until the enactment of the 1977 Clean Water Act, particularly section 404 of the act that requires a permit from the U. S Army Corps of Engineers for the physical alteration of any aquatic site (including wetlands), the wetland losses for North Carolina followed the national average and accounted for 49% (5,689,500 acres) of initial totals (Dahl 1990).

Significant wetland loss can be attributed to natural processes including sea level change (Hackney and Cleary 1987; White and Tremblay 1995) and storm derived erosion (Wray et al. 1995). Such losses have been magnified by anthropogenic influences (White and Tremblay 1995). The constant pressure related to growing populations along the coast has fueled increased development of bordering uplands resulting in subsequent anthropogenic degradation of estuarine habitats. As recently as 2005 application permits to the U. S. Army Corps of Engineers for wetland impacts totaled 31,141 acres, of which

20,754 acres were approved resulting in a required 56,693 acres of wetland mitigation which could be compensated for by: 1) permittee habitat creation, 2) mitigation banking or the purchase of credit in an existing or future restored (replacement of an existing habitat), created (creation of a new habitat) or enhanced habitat (modification of an existing habitat to improve it) and 3) in lieu monetary fee mitigation to compensate for the habitat lost (Zedler 1996).

Because of the increased awareness of estuarine function as CTZ's between uplands and marine environments (Levin et al. 2001), their significant ability to filter terrigenous pollutants and waste, habitat importance for the support of significant portion of commercially valuable fisheries and forage species (Hettler, 1989, Fonseca et al. 1990) and their critical linkage to fishery production (Boesch and Turner, 1984; Zimmerman and Minello, 1984), these habitats have been targeted by federal, state and non-government organizations for restoration and creation. Salt marsh restoration and creation has been a particular target for such activity due to the significant loss of these important habitats. Fewer logistic and monetary constraints associated with creating this habitat type compared to other less accessible habitat types, including submerged aquatic vegetation (Fonseca et al. 1994; Spurgeon 1998), oyster reefs (Mann 2005) corals and mangroves (Spurgeon 1998), also make salt marshes attractive restoration targets. However, salt marsh restoration is often restricted with regard to size and site placement. Monetary constraints of a project, which can range from $2,000-160,000/ha (Spurgeon 1998), will often determine project size, and logistical constraints often determine the project's site location, while habitat function aspects related to these factors are often

neglected. The resulting product of such creations are generally small (often < 4000 $m^2$) restorations consisting of monotypic stands of low salt marsh, comprised of smooth cordgrass (Spartina alterniflora) (Matthews and Minello 1994) that are situated at a site based on accessibility.

Because of high cost constraints associated with monitoring faunal use of restorations (Phinn et al 1996), limited monitoring has typically occurred (Matthews and Minello 1994). Monitoring that has been conducted consistently indicates unequal function of these habitats compared to reference habitats (Meyer et al. 1996; Rozas and Minello 2001), even after a decade of maturation (Meyer et al. 1996), leaving scientists to consider which factors are essential for constructions of comparable functioning. While investigators have conducted research to understand factors that might contribute to differential fish and decapod (nekton) use of salt marsh habitat, most of this work has targeted habitat level features. Recent work has targeted the importance of salt marsh edge versus interior use by nekton for both naturally occurring (Peterson and Turner, 1994; Baltz et al. 1993) and created salt marsh habitats (Minello et al. 1994; Rozas and Minello, 2001), effects of habitat heterogeneity (Meyer et al. 1996), the influence of hydroperiod (Rozas, 1995), the affect of creek branch order on nekton catch (Rozas and Odum, 1987; Rozas et al., 1988; Hettler 1989), the influence of vegetation stem density on potential predation (Heck and Thoman, 1981), and the influence of marsh topography and shallow water pools on resident species such as mummichog (Fundulus heteroclitus) (Kneib 1984; Talbot and Able 1984; Able and Hagen, 2000, Able et al. 2003).

Recent work has begun to focus on the influence of landscape level factors on

restored salt marsh function, including the importance of proximity to deep water (Rozas and Minello 2001) and movement of resident nekton among various sub-habitats comprising salt marsh habitats (Teo and Able 2003). Such efforts have been partly driven by awareness of the increasing size of mitigated (Grothues and Able 2003) and navigation related (Meyer 2001) restorations, and habitat improvement (Rozas and Minello 2001) related projects, and efforts to produce created habitats that have sufficient scale and habitat linkages as did those lost. However, no research has specifically examined the influence of salt marsh size or degree of isolation relative to like habitat. Both attributes could significantly influence faunal colonization and habitat function (MacArthur and Wilson 1967; Simberloff and Wilson 1969; Simberloff and Abele 1976; Wahlberg et al. 2002). A better understanding of how isolation and the size of salt marsh habitats relate to habitat function is critical for resource managers in understanding the influence of these factors on functioning of restored salt marsh, and the effects of habitat fragmentation on existing salt marsh habitat functions, as well as other estuarine habitats.

The proximity of a habitat to other similar habitats, regardless of size, and size of the habitat regardless of degree of isolation from like habitats, has been considered important for recruitment of new individuals and the number of species the habitat can support (Rieman and McIntyre, 1995; Acosta, 1999; Lockwood et al., 2002). Terrestrial research suggests that growth, production and resilience of populations with limited dispersal ability occupying small isolated habitat patches may be lower than that of similar-sized habitat with connectivity to expansive habitat, or to the larger habitat unit itself (Fahrig and Merriam, 1985). One implication of isolation is increased population

extinction risk for isolated resident species with limited dispersal ability (Fahrig and Merriam, 1985). While support for the theory of island biogeography and the affect of connectivity to similar habitats via dispersal ability of organisms has been demonstrated in freshwater environments (Rieman and McIntyre, 1995; Gotelli and Taylor, 1999), there are few examples within estuarine environments. However, similar relations may occur in estuarine habitat for species with limited dispersal ability, including killifish (Fundulus spp.), which dominate numerically in many salt marshes and have a limited home range (Lotrich 1975; Abrams 1985; Teo and Able 2003). Compared to contiguous salt marshes, the expanse of shallow and deep water habitats that surround island salt marshes may restrict immigration and emigration of marsh residents (those species for which the presence of marsh habitat is critical and in which they spend most of their life) due to increased predation vulnerability (Heck and Thoman, 1981), and isolate island marsh populations. The population size of species with limited dispersal ability that reside within isolated island marshes may be more adversely affected by lack of resources, including critical habitat types, than non-isolated habitat (MacArthur and Wilson 1967). Effects may be particularly evident for created salt marsh habitats which initially have simple habitat complexity and faunal diversity (Minello and Zimmerman, 1992; Sacco et al. 1994; Levin et al., 1996). Utilization of small isolated salt marshes by resident species might be lower than non-isolated salt marsh, and have populations with less resilience to ecological perturbations.

Recent work suggests that salt marsh populations for species with limited dispersal ability, including Fundulus heteroclitus, can be fished out at isolated salt

marshes while populations of more transient species with good dispersal ability, such as pinfish (Lagodon rhomboides), might be little affected (Meyer et al. 1996). However, current projects in areas throughout the USA have targeted the restoration of isolated island salt marsh habitats as primary candidates for habitat restoration using clean dredge material from navigation channels (Meyer 2001) in an effort to reduce anthropogenic impacts on restored habitat, improve avian (Rounds et al. 2004) and fisheries resources (Meyer 2001), and define legal ownership issues. This might result in low colonization rates for species with limited dispersal ability to these habitats. This might also lead to reduce mixing among populations within a geographic region, as might be predicted by metapopulation (Harrison and Taylor 1997; Wahlberg et al. 2002) and island biogeography theories (MacArthur and Wilson 1967; Simberloff and Wilson 1969; Simberloff and Abele 1976; Wahlberg et al. 2002), and lead to longer colonization times for these isolated habitats than less remote locations (MacArthur and Wilson 1967; Simberloff and Wilson 1969; Simberloff and Abele 1976). The function of these restored habitats, regardless of size, might then be temporally impaired. If this is the case, then consideration of site proximity to like habitat and size are important for effective restoration efforts.

Similarly, resident faunal populations with restricted dispersal ability, such as Fundulus heteroclitus, in isolated habitats might be particularly susceptible to local extinction if unfavorable conditions occur. Such a pattern has been noted for bull trout in fresh water stream habitats by Rieman and McIntyre (1995). Also, based on predictions of patch occupancy models, minimal habitat size thresholds might exist for sustained

occurrence of a species (Hanski 1994, Rieman and McIntyre, 1995). More transient and mobile estuarine species, such as Lagodon rhomboides (Hettler, 1989), would be predicted to show little local population variability within salt marsh habitat due to isolation or patch size.

A key factor when considering the creation of habitat, including intertidal salt marsh, should be whether or not that habitat acts as a source or a sink for target species populations. Does the size of a habitat influence the ability of species with poor dispersal ability to maintain itself? Estuarine salt marsh habitats might act in similar manner as terrestrial habitats for dispersal restricted species, and small habitats with little connectivity to like habitats might be prone to local extinctions (Harrison and Taylor 1997).

This dissertation research was conducted to assess the importance of landscape level habitat features, particularly habitat size and degree of isolation (distance) from similar habitats, on salt marsh resident nekton populations. Various methodologies were used to accomplish the research presented in the following chapters. Primarily emphasis was focused on examining recruitment, immigration, and predation risk (key variables in metapopulation and island biogeography models) for these target species among non-isolated mainland, isolated large island and small island salt marshes. Distribution estimates involved the use of baited eel pots, and examined habitat use specificity for Fundulus heteroclitus and Lagodon rhomboides. Nekton community structure for these three salt marsh types was examined using fyke and block net collections. Recruitment potential to these three marsh types for the target species was examined using Breder

traps and pit traps along with fyke and block net collections. Assessment of predator abundance (one measure of risk) was examined using gill net collections. Immigration and re-colonization assessment was accomplished by the removal of F. heteroclitus and L. rhomboides populations from large island salt marsh sites using baited eel pots and re-sampling the islands to detect population increases.

Specifically, Chapter 2 examines the nekton community structure at these sites related to associations with habitat features and the distribution of two nekton species Fundulus heteroclitus and Lagodon rhomboides, that are known to utilize and co-dominate in salt marshes along the North Carolina coast. Chapter 3 examines potential source/sink relations of the different salt marsh types with regard to F. heteroclitus and L. rhomboides recruitment and population maintenance, and predation potential by high order predators. Chapter 4 examines the colonization potential for F. heteroclitus and L. rhomboides to isolated salt marsh islands.

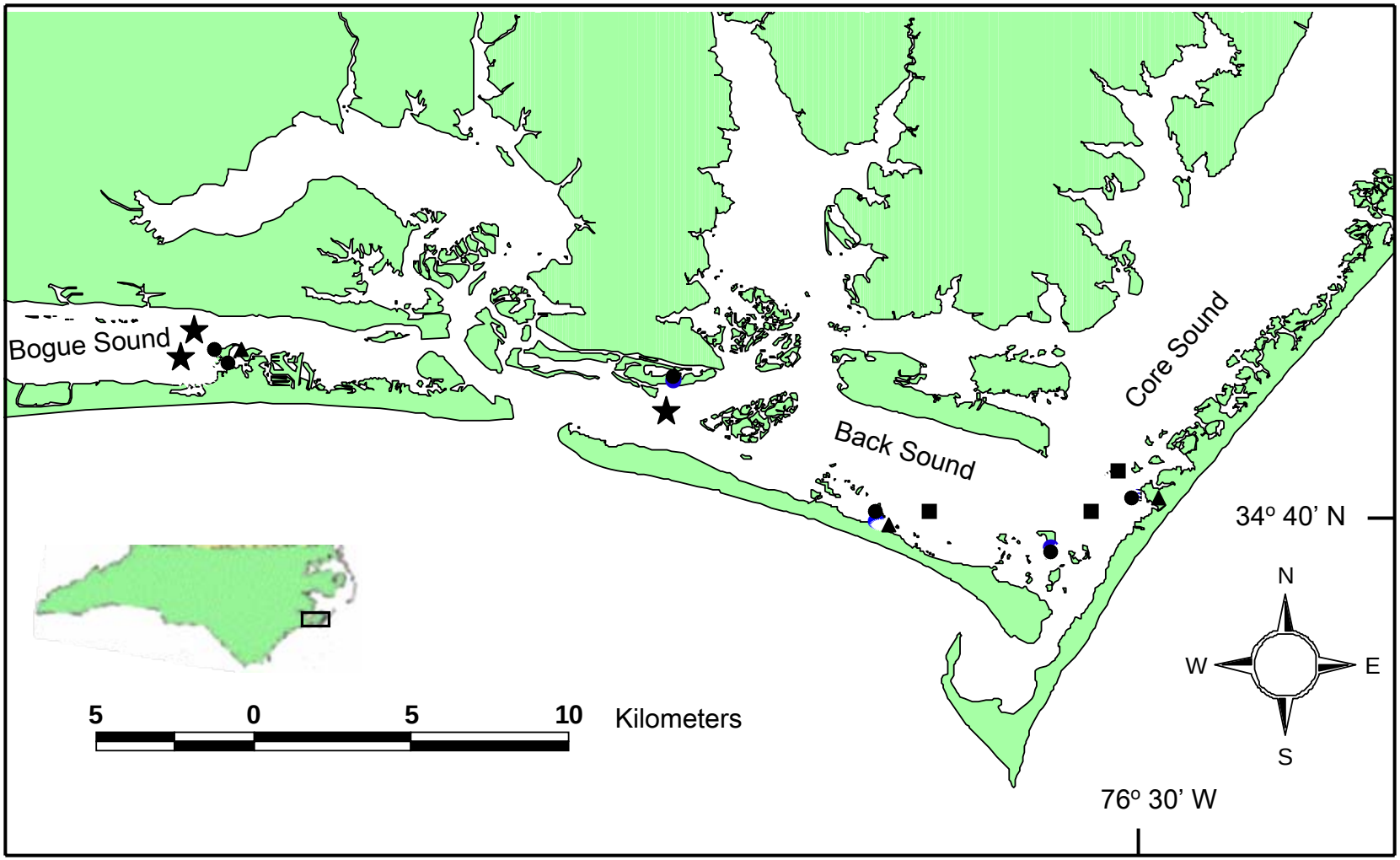
To examine both size and isolation effects six naturally occurring island salt marshes, three each of two discreet size categories: small (~400-1000 $m^2$) and large (~3,000-10,000 $m^2$), were paired with six mainland salt marshes (> 76,000 $m^2$ in size). All six pairs were located amongst shallow water flats composed of a seagrass and unvegetated bottom mosaic within Bogue, Back and Core Sounds of North Carolina, USA (Fig. 1-1). A pair consisted of isolated island and non-isolated mainland salt marshes within close geographic proximity (within ~1.0 km) of one another. Site selection criteria for each pair included observed similarities in: vegetation at equivalent tidal elevation, similar salinity and topographical slopes, and lack of dendritic rivulet

development. Island salt marsh sites were typically at least 400 m away from the nearest other salt marsh. This distance was beyond the maximum reported summer dispersal range for Fundulus heteroclitus (Lotrich, 1975; Teo and Able 2003).

Examination of the estuarine distribution of two representative fish species using eel pots (Chapter 2) involved comparison among mainland and all island salt marshes (large and small island types were combined to represent isolated salt marsh) and a pair consisted of isolated island (~400 -10,000 $m^2$ in size) and non-isolated mainland salt marsh within ~1.0 km of one another. In addition to these paired sites, eel pot collections also included three interior salt marsh sites, located adjacent to three mainland salt marsh sites, one each in Bogue, Back and Core Sounds, which were similarly sampled. Interior salt marsh sites were located near the headwaters of salt marsh creeks which bisected the interior salt marshes (Figure 1-1). These interior salt marshes were also used for larval recruitment estimates for Fundulus heteroclitus (Chapter 3) along with the previously mentioned paired small, large and mainland salt marsh sites. Re-colonization estimates (Chapter 4) utilized only the large island salt marshes (all located within Back and Core Sounds) and a nearby mainland salt marsh (located in Core Sound).

The primary goal of this research is to better understand the affect of salt marsh habitat size and location on overall habitat function. The combined efforts of these empirical studies will provide resource managers a better understanding of the functional aspects of estuarine habitats in order to better manage our estuarine ecosystems for future generations to enjoy.

Fig. 1-1. Site locations within Bogue, Back and Core Sounds of North Carolina. Locations corresponding to the circles are mainland salt marsh sites, squares are large island salt marsh sites, stars are small island salt marsh sites and triangles are interior salt marsh sites.


Bogue Sound
Back Sound
Core Sound
34° 40' N
76° 30' W
N
W
E
S
5
0
5
10
Kilometers

# CHAPTER 2.

# A COMPARISON OF NEKTON UTILIZATION AND DISTRIBUTION AMONG ISOLATED ISLAND SMOOTH CORDGRASS (SPARTINA ALTERNIFLORA) MARSH AND MAINLAND SALT MARSH.

# INTRODUCTION

The realization of significant wetland loss, including emergent salt marsh, within the continental USA (Dahl 1990) has created an increased awareness of the importance of these habitats and fueled efforts to preserve and restore them through government mandated mitigation (Zedler 1996) and non-government organization efforts. While such efforts are noble, logistical constraints and the high cost of restoring emergent salt marshes (Zedler 1996) has often resulted in the production of small (often < 4000 $m^2$) patches consisting of monotypic stands of low salt marsh, comprised of Spartina alterniflora (Matthews and Minello 1994), placed in locations based on logistical and cost consideration rather than the proximity to like habitat or other landscape level features. However, the proximity of a habitat to other similar habitats, regardless of size, and size of the habitat regardless of degree of isolation have been theorized to be important in terms of potential recruitment of new individuals and the number of species that can be supported (MacArthur and Wilson 1967; Rieman and McIntyre 1995; Acosta 1999; Lockwood et al. 2002). A recent study (Meyer et al. 1996) of isolated created salt marshes also indicates that local populations of nekton with poor dispersal ability, such as Fundulus heteroclitus (Lotrich 1975; Abrams 1985), can be driven to extinction by collection without replacement, underscoring the importance of isolation to population dynamics of these species. Conversely populations of more transient species with good dispersal ability, such as pinfish (Lagodon rhomboides), might be little affected (Meyer et al. 1996).

Salt marsh restoration projects throughout the USA have increasingly targeted the restoration of island habitats as primary candidates for habitat restoration (Meyer 2001)

to reduce anthropogenic impacts on restored habitat, enhance avian and fisheries resources, and define legal ownership issues. Potential colonization of these islands by species with limited dispersal ability might be low, as predicted by metapopulation (Harrison and Taylor 1997; Wahlberg et al. 2002) and island biogeography theories (MacArthur and Wilson 1967; Simberloff and Wilson 1969; Simberloff and Abele 1976; Wahlberg et al. 2002), or take longer for colonization than less remote locations (MacArthur and Wilson 1967; Simberloff and Wilson 1969; Simberloff and Abele 1976). Based on research from terrestrial environs, growth, production and resilience of populations with poor dispersal ability within small isolated habitat patches may be lower than that of similar sized areas with connectivity to expansive habitat (Fahrig and Merriam 1985). These factors suggest a high potential for extinction of isolated resident populations of species with poor dispersal ability (Fahrig and Merriam 1985). While island biogeography theory and importance of connectivity has been supported by work performed in terrestrially-influenced freshwater environments (Rieman and McIntyre 1995; Gotelli and Taylor 1999), there have been few attempts to do so within the estuarine environment. Similar relations may occur in estuarine habitats for species with poor dispersal ability and that have limited home range.

Resident faunal populations with restricted dispersal ability, such as Fundulus heteroclitus, in isolated habitats might be particularly susceptible to extinction events, as has been noted for fish populations in fresh water streams (Rieman and McIntyre 1995) and forest rodent populations (Fahrig and Merriam 1985). Also, based on patch occupancy theory, larger habitats should be able to sustain larger populations and may be critical to maintain threshold sustainable densities of resident species with poor dispersal

ability (Rieman and McIntyre 1995). More transient and mobile estuarine species, such as Lagodon rhomboides (Hettler 1989), would be predicted to show little local population variability within habitat patches due to isolation or patch size.

When considering the creation of habitat, including intertidal salt marsh, a key factor should be whether or not that habitat acts as a source or a sink for target species populations. Does habitat size influence its ability to support certain target species? One might expect that estuarine salt marsh habitat might act similar to terrestrial habitats for dispersal restricted species, and that small habitats with very little connectivity to like habitats might be prone to local extinctions and exhibit a 'blinking light' pattern of utilization (Harrison and Taylor 1997). To gain insight into the potential influence that habitat size and location might have on basic functions, this study examined existing natural 'small' and 'large' isolated islands and expansive non-isolated 'mainland' salt marshes to serve as proxies for created habitats. The objectives of this study were to: 1) examine nekton community composition, diversity, and abundance based on salt marsh size; 2) compare and contrast utilization patterns for nekton with poor dispersal ability versus those with good dispersal ability; 3) examine the population distribution of two representative species, F. heteroclitus and L. rhomboides, within coastal estuarine shallow water flat and intertidal salt marsh habitats; and 4) examine environmental, biological and physical factors (in particular salt marsh size and degree of isolation from like habitat) that might be responsible for the observed species abundance and distribution patterns. Naturally occurring salt marshes were used because the ultimate goal of restored salt marsh is typically to assume the functional characteristics of naturally occurring salt marshes; and because of the difficulty in producing restored salt

marshes in the appropriate settings; the unpredictability of producing stable restored salt marshes; and inherent problems with time lags to maturity for restored salt marshes.

## MATERIALS AND METHODS

### Sites

The six naturally occurring island salt marshes that were paired with six mainland salt marshes and the interior salt marshes for the various collections (see Chapter 1) were located amongst shallow water flats composed of a seagrass and unvegetated bottom mosaic within Bogue, Back and Core Sounds of North Carolina, USA (Fig. 2-1). Island salt marsh sites were typically at least 400 m away from the nearest other salt marsh.

### Physical Parameters

Salt marsh vertical range was measured at each site using a laser level and stadia rod with a detector sensor (accuracy = $\pm$ 5 mm at 310 m) (Meyer et al. 1997). The vertical distance between the lowest elevational occurrence of salt marsh vegetation and the highest point measured at a site was considered to be the vertical range for a site.

Area of the salt marsh sites was estimated using a sub-meter Global Positioning System (GPS). This GPS unit was used to delineate total salt marsh area of each island salt marsh as well as low salt marsh (salt marsh typically flooded during diurnal neap tides) and shallow salt marsh (salt marsh habitat flooded only during spring or astronomically high tides and observed to contain high marsh vegetation species) sub-habitats. Vegetation (occurrence of high marsh species including Spartina patens, Salicornia virginica, Distichlis spicata, etc.) and elevation breaks were used to delineate

shallow and low salt marsh sub-habitats. Mainland salt marsh habitat and sub-habitat size is presented as minimal values based on the smallest mainland site sampled due to the relatively large size and interconnectedness of the mainland salt marshes used. Island low salt marsh area and low salt marsh access perimeter were delineated using GPS to examine nekton abundance related to area/perimeter estimates. Mean breadth estimates of the low salt marsh (measured at 10 m intervals along the low marsh edge, from the marsh edge to the shallow marsh boundary or island center if shallow marsh was not present) were obtained for each island and compared to nekton abundance. Distance from island salt marshes to the nearest other salt marsh was estimated using GPS while distance for mainland sites was considered to be zero due to their connection with adjacent areas.

Sediment Analysis

Three 3.8 cm diameter x 3.0 cm deep sediment cores were randomly taken within adjacent flat and the shallow and low salt marsh portions of each site. All cores were analyzed for organic content and particle size composition.

Percent organic matter for each core fraction was estimated by ashing at 500$^{o}$ C for 24 h, after sediment had been dried at 100-110$^{o}$C to a constant weight (24-36 h). Percent organic matter was determined through comparison of the post- and pre-ash weights.

Wet sieving techniques, following methods used by Murphey and Fonseca (1995), were performed to determine the particle size fractions of the samples, delineating gravel (including shell hash), sand, and silt-clay fractions. A 20 g sample

was treated with 5 ml of wetting solution (40 g/L sodium laurel sulfate or sodium hexametaphosphate) and the mixture was hand-blended. This slurry was poured through a 2 mm and a 0.063 mm sieve series. Each core fraction was washed until the wash water was clear (1 to 2 L of water) and the effluent was collected. Fractions contained in the 2.0 mm and 0.063 mm sieves were collected. Detrital material was removed from the fraction collected in the 2.0 mm sieve. Filtrate was thoroughly mixed and three aliquots were obtained for silt-clay estimates. The remaining effluent was measured volumetrically and 75 ml (representing the three aliquots) was added to that amount to represent total effluent. All collected material was dried at 100-110ºC to a constant weight (24-36 h). The dry weight of the silt-clay fraction was determined by: ((mean g/aliquot)/(25 ml/aliquot)) X total vol. effluent = g silt-clay fraction/sample (Murphey and Fonseca 1995). Percent silt-clay, sand and gravel for each core fraction was estimated based on the total dry weight of each core fraction.

Marsh Vegetation

At each site, vegetation stem density counts and morphometrics, based on methodology used by Cowie et al. (1992) and Meyer et al. (2001), were measured during the fyke net collection periods. To account for vegetation zonation within each of the 10 m wide nekton collection areas, each marsh site was divided into two equal portions, a lower marsh section which encompassed an area from the lower marsh fringe up to one-half the distance to the back of the site, and the upper marsh section which encompassed the remainder of the area from the back of the site to one-half the distance to the front of the site. Within each site four randomly selected vegetation count locations were

located; two within the lower marsh section and two within the upper section. Within each count location the number of live and dead plant stems within a 0.25 $m^2$ quadrat were recorded for each plant species.

Stem height and diameter were estimated for each quadrat by measuring the height and diameter for up to 10 live and dead stems (diameter was measured at 10 cm height) of each macrophyte species encountered within the quadrat. Stems were chosen based on closest proximity to a pre-determined quadrate corner. Areal stem coverage was then estimated for each quadrate by multiplying the mean stem diameter for each macrophyte species by the number of stems observed for that species.

Marsh Nekton Collection

During 2002-2003 (year one) and 2003-2004 (year two), nekton were collected within each salt marsh site to examine utilization patterns among the different salt marsh types (especially for the two target species, Fundulus heteroclitus and Lagodon rhomboides) during different life history stages. Collections occurred during late spring/early summer (June), a time of peak young of year (YOY) recruitment for F. heteroclitus, and a period when L. rhomboides have attained juvenile size class status; the mid fall (November), a time when YOY F. heteroclitus have attained juvenile or adult size and L. rhomboides have attained adult size; and late winter/early spring (March), a time when L. rhomboides YOY recruit to estuarine habitats and adult F. heteroclitus prepare to spawn within the marshes. Fyke and paired block nets constructed of black 3.2 mm mesh were used for marsh faunal collection (McIvor and Odum 1986; 1988; Meyer et al. 1996; 2001). Fyke nets with a 1 $m^2$ mouth and 5.5 m wings, in combination

with 30 m block nets, were used at each site to collect marsh nekton. At each site, 10 contiguous linear meters of marsh fringe were demarcated for next day sampling. On the day prior to nekton collections, paired sets of fyke/block net attachment poles were set 10 m apart along the marsh fringe and were paired with back block net poles set at the highest elevation point for the marsh site (10 m back into the interior of the marsh for some sites, 24 m at other sites, depending on marsh topography). One end of each block net (up to 30 m long) was attached to a back block net pole and bundled to it in preparation for marsh fauna collection. A ten meter long back block net was also connected to the paired back block net poles at each site. This was also bundled to one of the back block net poles in preparation for nekton collection and strung across the back of the demarked fyke net area during collections to cordon off the movement of nekton into and out of the area sampled. During site preparation, debris which might hinder lead line set on the sediment surface was removed from the areas where nets were deployed. Preparation also include the connection of each block net top to guidelines, which were strung between the fyke/block net poles, and back block net poles and between back block net poles. Once a site is prepared, it was allowed to sit for at least one complete tidal cycle prior to sampling. Fyke and block nets were deployed at a site during a morning high tide. During deployment a consistent approach was used in which first the fyke net was set in place, followed by simultaneously walking the side block nets and finally the back block net into position to cordon off the sample areas. Nekton were then collected once the tide evacuated from the fyke nets during the subsequent low tide, and the collection area was surveyed for nekton stranded on the marsh surface. Nekton were identified to species, measured on site and released live. For each site the number of

individuals and wet-weight biomass for each species were recorded. All of the individuals for a species were measured or, if numerically abundant, a randomly selected subsample of at least 100 individuals, or 5 % of the total (whichever was higher), were measured (standard length for fishes, total length for shrimps and carapace width for crabs).

Salinity (as measured with a temperature compensated refractometer, accuracy of 0.1 ppt) and water temperature were measured for fyke and block net collections at each site during each collection period. During each fyke and block net collection period relative marsh elevations were also measured at the front fringe and back block net stakes of each site using the water surface as a level (Meyer 1994). Annual site averages were calculated for each parameter per site.

Distribution Assessment

During 2003-2004 high and low tide distributions and relative abundances of Fundulus heteroclitus and Lagodon rhomboides were examined using eel pots (Halpin 1997; 2000; Kneib and Craig 2001) set at each of the salt marsh sites semi-monthly. Baited eel pots were used for these collections because of their potentially higher catch attraction (Reebs et al. 1995) and retention (Whitelaw et al. 1991) compared to unbaited eel pots. Baited eel pots were preferable to unbaited eel pots because the objective of this assessment was to determine if F. heteroclitus and L. rhomboides did in fact utilize locations within salt marshes and adjacent shallow water flats at high tide, and whether or not shallow water flat habitats adjacent to salt marshes acted as less hospitable matrix habitat. Further, a second objective was to determine distances that F. heteroclitus and L.

rhomboides might venture into adjacent shallow water flats from salt marsh habitats during low tides, and the higher attraction of baited over unbaited eel pots would increase potential encounter for capture. Eel pots were 80 cm in length, 22.5 cm in diameter, were constructed of 0.5 cm bar mesh and had conical capture ends that were positioned inward with 6 cm long by 3 cm wide capture openings (Halpin 1997; 2000; Kneib and Craig 2001).

High tide distributions were examined by eel pots set at 100, 25 and 5 m seaward of the salt marsh edge in the shallow water flats that bordered island and mainland salt marsh sites, at a mid point (MP) between the paired island and mainland salt marsh sites, and at the salt marsh edge (ME), 2 m inside of the salt marsh edge (-2 m) and within the shallow salt marsh (SM) portion of the site for the island, mainland and interior salt marsh sites. In addition to these locations, eel pots were set within a deep area of the salt marsh creek (MC) that fed interior salt marsh sites. For high tide collections eel pots were baited with 210 g of dry dog food and allowed to fish for a short duration (approximately 1.0 h) (Kneib and Craig 2001) within a three hour time window, 1.5 hours before to 1.5 hours after high tide. Relative abundance was examined based on catch per hour (CPH) fished for each eel pot (Kneib and Craig 2001).

Low tide distribution for Fundulus heteroclitus and Lagodon rhomboides were sampled for locations that would not be exposed during a typical low tide (Halpin 1997; 2000). These locations included mid-point (MP), 100 and 25 m locations seaward of the salt marsh edge for the island and mainland salt marsh sites, and the salt marsh creeks for the interior salt marsh sites. Due to logistic constraints, low tide distribution assessments for F. heteroclitus and L. rhomboides were begun during ebb tides and were collected

during the following tidal flood (~10 hours later). Because eel pots at the MP, 100 and 25 m and marsh creek locations were fished a complete tidal cycle during the ebb tide distribution, CPH estimates were not used in the analyses due to increased escape potential associated with prolonged soak times (Whitelaw et al. 1991; Kneib and Craig 2001). Instead of CPH, catch estimates for low tide comparisons involved presence (1) and absence (0) information. These same criteria were used for high tide distributions when compared to these ebb tide catch observations. During the ebb tide distribution assessment each eel pot was baited with 315 g of dry dog food, to insure that the bait supply was not exhausted prior to collections and to improve catch retention (Whitelaw et al. 1991).

For each eel pot, nekton were identified to species enumerated and measured. If numerically abundant, a randomly selected subsample of at least 30 individuals for each fish species were measured (standard length). Individuals collected were released live back at the point of collection.

Salinity (as measured with a temperature compensated refractometer, accuracy of 0.1 ppt) and water temperature were measured at each site during eel pot collections, along with relative eel pot water depths, as measured during set and collection (Meyer 1994). Eel pot set and collection times were recorded for catch per hour (CPH) estimates. Annual site averages were calculated for each parameter per site.

## Statistical Analysis

Because of the preponderance of zero catches, comparisons for the different salt marsh types, sample locations and time periods utilized the Kruskal-Wallis test, a

conservative distribution-free single classification method of analysis (Sokal and Rohlf 1981), to test for distributional differences in nekton abundance. During each eel pot collection period, Fundulus heteroclitus and Lagodon rhomboides size was also compared among island, mainland and interior salt marsh types using the Kruskal-Wallis test (Sokal and Rohlf 1981) with the capture of each individual considered to be an independent event (Kneib and Wagner 1994; Kneib and Craig 2001) not each eel pot (Kneib and Craig 2001). For size comparisons, fish collected at the marsh edge and at 2 m inside of the marsh edge were pooled and considered to be low salt marsh due to the similarity of habitat type fished. The Kruskal-Wallis test was also used to compare salt marsh vegetation parameters including stem density counts and estimates of areal stem coverage for each salt marsh type and physical parameter averages measured for each site and sample location during each year. For sediment analysis shallow and low salt marsh and mudflat collection regions were averaged together per site and the recorded proportions were arcsine transformed prior to regression analysis, and among-salt marsh type comparisons using analysis of variance (ANOVA) and post hoc Student Neuman-Keuls tests. Replication level was the site; that area encompassed within the fyke net collections.

Principal Components Analysis (PCA) examined the relationship between cumulative nekton abundance at each salt marsh site with other measured physical and biological factors to identify the variables most responsible for the observed standardized data variance (Johnson and Wichern 2002). Only island sites were included in the analysis owing to that fact that only the island sites had precise measures of total, low and shallow salt marsh areas for each site. An eigenvalue of 1.0 was the minimum value

limit for a principal component to be considered important towards accounting for data variance (SAS 1987; Johnson and Wichern 2002). Stepwise Multiple Linear Regression Analysis (SMLRA) followed PCA to determine which factors identified by PCA contributed most towards explaining observed cumulative abundances of common nekton species and nekton groups (Sokal and Rohlf 1981). A maximum p value of 0.15 was set for variables to be included in models with a maximum $p = 0.05$ level for the overall model to be considered significant. Linear regression analysis (regression analysis) was utilized to examine the factors that best described a linear relationship with individual nekton species and nekton group abundance, and to examine Fundulus heteroclitus and Lagodon rhomboides eel pot CPH compared to water depth. Regression analysis also tested for significant trends between mean size of F. heteroclitus and L. rhomboides per eel pot and water depth using combined collections from the interior and mainland salt marsh habitats. These two 'mainland' salt marsh types were combined in this size-water depth regression analysis based on similarities in functional habitat connectivity. For eel pots to be used in the size-water depth analysis the minimum number of individuals collected for a species per eel pot had to be $\geq 3$. Data were included in PCA after ln $(x + 1)$ transformation (SAS 1987; Fonseca and Bell 1998) and arcsine transformation for proportional data (Sokal and Rohlf 1981; SAS 1987). For SMLRA and regression analysis non percentage data were tested for normality using the Shapiro-Wilk test (Shapiro and Wilk 1965; Sen et al. 2003). If these data were found not to be normal, data were ln $(x + 1)$ transformed and again tested to assure data conformity.

Nekton community composition similarity was tested for each salt marsh type to determine if differences existed among the different salt marsh types using Analysis of

Similarity (ANOSIM) (Clarke and Warwick 2001) based on the Bray-Curtis Similarity Index, and similarity percentages (SIMPER) to determine which species contributed most to observed difference. Non-metric multidimensional scaling ordinations (nMDS) were generated using square root transformed data (Clarke and Warwick 2001). Square root transformation was sufficient to conform data to two dimensional ordination with acceptable data stress levels (data conformity to normal distribution) for analysis (stress $\leq 0.2$). Differences in community structure similarity (represented by R) between salt marsh types were tested using one factor ANOSIM with significance level $p = 0.05$. For all test statistical analyses minimal significance level was $p = 0.05$.

## RESULTS

### Physical Parameters

As selected for, consistent physical patterns were present among salt marsh types including habitat dimension characteristics and location (Table 2-1). The amount of total, low and shallow salt marsh areas, which encompassed the sites, were significantly different among mainland, large island and small island salt marsh types (Table 2-1). While large and small island salt marsh types were significantly farther away from like habitat than mainland salt marshes, there was no significant difference between distance associated with island salt marsh types (Table 2-1). The measured vertical range of the salt marsh for the different salt marsh types indicated large and small island salt marsh types had significantly wider ranges of vegetation occurrence than that for mainland salt marshes (Table 2-1). While a trend for higher sediment organic and silt-clay content was observed, with the mainland containing the highest levels and the small island salt

marshes the least, significant differences were only observed between the mainland and small island salt marshes for percent silt-clay sediment composition (Table 2-1). Salinity measured within the salt marsh types tended to be relatively consistent among years and salt marsh type. Water temperature at the island salt marsh sites varied more than that at the mainland sites among years and among salt marsh island type (Table 2-1).

For eel pot collections, salinities and temperatures measured did not significantly differ between marsh types (Table 2-1). Measured water depths for the eel pots in general tended to show a pattern of slightly, though not significantly, deeper water depth within the mainland compared to the interior and the island compared to the mainland sites for most comparable distance locations. However, low salt marsh habitat (salt marsh edge and 2 m within the salt marsh fringe locations) within the interior salt marshes were observed to be significantly shallower (26.5 cm) that those of mainland (38.8 cm) and island (43.2 cm) salt marshes (Table 2-1).

Marsh Vegetation

Smooth cordgrass (Spartina alterniflora) dominated the salt marsh vegetation in areas sample for all salt marsh types during this study. Other species were relatively rare with the exception of glasswort (Salicornia virginica), which was infrequently observed within mainland and large island salt marshes (Table 2-2). While mean total vegetation and S. alterniflora stem density was consistently lower at large island salt marshes compared to both mainland and small island salt marshes for both years as a whole, no significant differences were observed (Table 2-2).

## Marsh Nekton Collection

A total of 49 nekton species were collected during the two year duration of this study. The highest total number of nekton species encountered during each year was consistently observed within the mainland salt marsh (35 and 31, yrs. 1 and 2, respectively), followed by the large island (31 and 30, yrs. 1 and 2, respectively) and finally the small island salt marsh (25 and 24, yrs. 1 and 2, respectively). Mainland and large island salt marshes consistently had more nekton species (mean number per salt marsh type per date and year) than the small island salt marshes. Mainland and large island salt marshes alternately had the highest mean number of nekton species throughout this study, while small island salt marshes consistently supported fewer nekton species (Figs. 2-2a-b ). Significant differences were observed between mainland and large island salt marsh types compared to small island salt marshes during March of year one and on average for year one as a whole (Figs. 2-2a-b).

When the total number of different nekton species encountered during the two year study duration was compared with salt marsh habitat size (total salt marsh, low salt marsh or shallow salt marsh), no relationship was apparent between the total number of species nor transient nekton species. However, the number of resident salt marsh nekton species per site was significantly related to total salt marsh area, low salt marsh and shallow salt marsh (linear regressions), with the strongest linear regression based on total salt marsh area (total island size) (Figs. 2-3a-c).

## Abundance Comparisons

Abundance patterns for most species were consistent during both year one and

two. Eight of the 23 most abundant nekton species consistently had highest density within mainland salt marsh habitats with density decreasing based on salt marsh size. Resident salt marsh species such as sheepshead minnow (Cyprinodon variegatus), Fundulus heteroclitus, naked goby (Gobiosoma bosc), rainwater killifish (Lucania parva), as well as transient species including striped mullet (Mugil cephalus), mojarra (Eucinostomus sp.), grass shrimps (Palaemonetes intermedius) and (Palaemonetes pugio) all showed this trend with significant differences apparent during at least one collection period or the year as a whole for C. variegatus, F. heteroclitus, M. cephalus, and P. pugio (Table 2-3). A contrary pattern was observed for four transient nekton species with a trend of higher density within small island salt marshes and density decreasing with salt marsh size. Lesser blue crab (Callinectes similis), striped killifish (Fundulus majalis), Atlantic silverside (Menidia menidia) and gulf flounder (Paralichthys albigutta) all showed this trend with significant differences apparent during at least one collection period or the year as a whole for all but F. majalis (Table 2-3).

Principal Component Analysis

Principal component analysis (PCA) included all nekton abundances for island sites only and contained 11 dependant variables. This analysis revealed that four principal components at the 1.0 eigenvalue threshold described ~94% of the standardized variance (Table 2-4). The first PC accounted for ~49% of the standardized variance with total marsh area, low marsh area, water temperature, high marsh area, stem diameter and marsh vertical relief having the highest loadings for this eigenvalue. The second PC accounted for ~18% of the standardized variance with sediment organic and silt-clay

contents having the highest loadings for this eigenvalue. The third PC accounted for ~15% of the standardized variance with salinity and nekton abundance having the highest loadings. The fourth PC accounted for ~11% of the standardized variance with vegetation canopy height, nekton abundance and salinity having the highest loading for this eigenvalue (Table 2-4)

Stepwise Multiple Regression, Linear Regression and Physical Factors

Stepwise multiple regression analyses revealed significant regression models for 15 of the 28 species and nekton groups tested (Table 2-5). The highest model $r^2$ values were observed for Callinectes sapidus and Menidia menidia with 1.00 and the lowest value observed was that for Palaemonetes pugio with 0.7455 (Table 2-5). For C. sapidus total salt marsh area loaded first into the model, while mean water temperature loaded first for M. menidia. For P. pugio the only factor that loaded was sediment silt-clay content. Of the 13 species and groups with significant regression models, eight of the 13 loaded total or high salt marsh area size into the regression model first. These species and groups included C. sapidus, C. variegatus, and Mugil cephalus, for which total salt marsh area loaded first and Callinectes similis, Fundulus heteroclitus, Fundulus majalis, Lucania parva and resident nekton for which high salt marsh area loaded first (Table 2-5). For the five other species with significant regression models, three had mean water temperature load first, M. menidia, Palaemonetes vulgaris, and pink shrimp (Farfantepenaeus duorarum), while salinity loaded first for Lagodon rhomboides and sediment silt-clay content for P. pugio (Table 2-5).

Linear regression analysis for individual parameters indicated only high salt

marsh area significantly influenced the cumulative abundance of Fundulus heteroclitus, all resident marsh nekton and Fundulus majalis. However, the relationships for F. heteroclitus and resident marsh nekton were opposite that of F. majalis with increase in cumulative abundance observed for F. heteroclitus and resident marsh nekton compared to shallow salt marsh area and decreased cumulative abundance for F. majalis. Negative significant relationships between Callinectes sapidus cumulative abundance and total salt marsh area, and a lesser extent to low salt marsh area, were also observed (Figs. 2-4a-f). A significant negative relationship was observed between cumulative Lagodon rhomboides abundance and mean water salinity (Fig. 2-5). Though tested, no significant relationships were observed between low marsh area/perimeter ratios nor mean low marsh breadth and cumulative abundance for nekton collected.

Similarity Analysis

Strong seasonal patterns (stress > 0.2) required data separation by seasons, while combining years per season allowed reasonable data stress levels ($\leq 0.2$) for data ordination and ANOSIM. Analyses indicated that during June the most substantial differences in nekton composition between salt marsh types occurred and involved significant differences ($p \leq 0.05$) among mainland, large and small island salt marshes with the least differences apparent between mainland and large island salt marshes (Table 2-6, Fig. 2-6a). During both November and March collections, no significant difference in nekton composition was observed between the mainland and large island salt marshes. Significant differences were apparent between mainland and small island salt marshes, and large island compared to small island salt marshes, with the comparisons between

large island versus small island salt marshes observed to be more dissimilar (Table 2-6, Figs. 2-6b-c). Six species contributed the most to distinctions between salt marsh types (> 10%), with Lagodon rhomboides substantially contributing to all comparisons during each season while Fundulus heteroclitus, Fundulus majalis and Menidia menidia substantially contributed to over half of all comparisons (Table 2-7).

Distribution Comparison and Assessment

Eel pot collections revealed relatively consistent patterns of habitat use for both Fundulus heteroclitus and Lagodon rhomboides. While zero and low catches prevented the observation of trends and distribution patterns for both species during January 2004 and March 2004 and data for these collection periods are not presented, catches during the other collection periods indicated habitat distributional differences among the two species and size dependant habitat use.

During high tide collections, F. heteroclitus were exclusively collected within the salt marshes, including the marsh edge (ME), -2 m and shallow marsh (SM) for island, mainland and interior salt marsh types, and within the marsh creek (MC) of interior salt marsh types (Table 2-8). Among the ME, -2 m and SM locations, F. heteroclitus CPH was consistently higher within the SM followed by -2 m and finally ME locations for the mainland and island salt marsh sites (Table 2-8). Within the interior salt marshes, a CPH pattern shift was apparent based on temporal period. Only during September 2004 was a significant difference for CPH observed among locations for interior salt marshes, and also the only time period in which the CPH pattern for interior salt marsh was similar to those observed for both mainland and island salt marsh sites (Table 2-8). Generally, low

salt marsh areas and intermediate distance shallow water flat locations had the highest CPH of Lagodon rhomboides, including 100 m and 25 m for shallow water flats, and ME and -2 m locations for salt marsh habitats, particularly during summer collections. Mid-point (MP), 5 m and SM locations tended to have lower CPH compared to other locations for mainland and island salt marsh sites (Table 2-8). For interior marsh sites, while SM location consistently had lower comparative CPH, no significant differences in catch and location were apparent during any collection period.

Linear Regression Eel Pot CPH/Water Depth

Using regression analysis that included all distances, significant negative linear regressions were observed for Fundulus heteroclitus for individual salt marsh types and for all salt marsh types combined (Table 2-9). When comparing CPH for L. rhomboides to water depth, including all distances, significant negative linear regressions were observed for mainland and island salt marsh types, and for all salt marsh types combined. However, within interior salt marshes significant increases in L. rhomboides CPH with increasing water depth was observed (November 2003) (Table 2-9).

Comparing Fundulus heteroclitus CPH relative to water depth for within-salt marsh locations only, significant negative linear regressions were observed for individual salt marsh types and for analyses containing all salt marsh types combined. However, during May 2004 a significant positive F. heteroclitus CPH with increasing water depth relationship was observed for interior salt marshes (Table 2-9). When examining CPH for L. rhomboides, based on water depth, using analyses that included within salt marsh locations only, significant positive relations between L. rhomboides CPH and increasing

water depth were observed for mainland and island salt marsh types and for all salt marsh types combined (Table 2-9).

Size Distribution

Analysis of Fundulus heteroclitus and Lagodon rhomboides MSL per eel pot to water depth, used the combined catch of interior and mainland salt marshes (non-isolated salt marshes), while scarcity of F. heteroclitus at the island salt marshes precluded inclusion of data collected from these sites. This analysis revealed significant positive regressions for both species relative to fish MSL and average eel pot water depth (Fig. 2-7). During the four collection periods when individuals for both species were abundant, significant positive linear regressions were observed for F. heteroclitus during all four collections periods and two of four collection periods for L. rhomboides (Fig. 2-7).

Size differences between salt marsh types based on comparable locations were evident for both Fundulus heteroclitus and Lagodon rhomboides (Table 2-10). Generally, a trend of larger fish MSL for both F. heteroclitus and L. rhomboides was observed per location for island compared to interior and mainland salt marsh sites and larger MSL for mainland compared to interior salt marsh sites (Table 2-10).

Ebb Tide Distribution

Ebb tide occurrence of Fundulus heteroclitus and Lagodon rhomboides for the different distance locations based on time of year, revealed higher occurrence of F. heteroclitus at mainland 25 m and 100 m locations during November 2003 and March 2004 compared to any other collection period, with significantly higher occurrence

observed at the 25 m locations during November 2003 and March 2004 compared to both May 2004 and September 2004. For L. rhomboides, a consistent significantly higher occurrence was apparent at 25 m and 100 m locations for both mainland and island salt marsh sites during November 2003, May 2004, July 2004 and September 2004 compared to January 2004 and March 2004 (Table 2-11). While F. heteroclitus ebb tide occurrence at interior salt marsh creeks did not show significant differences between time periods sampled, L. rhomboides patterns demonstrated a trend similar to both mainland and island marsh sites with higher occurrence observed during the November 2003, May 2004, July 2004 and September 2004 compared to both January 2004 and March 2004 time periods (significant during the September 2004 time period) (Table 2-11).

## DISCUSSION

It was apparent that salt marsh size influenced nekton habitat suitability and that minimal total salt marsh or sub-habitat area size thresholds exist for certain nekton species. Habitat size related immigration/extinction ratios can directly influence the number of species habitats can sustain (MacArthur and Wilson 1967; With and Crist 1995) and might be a major contributor to this pattern. The size of adjacent habitats and sub-habitats can also influence the population density for rare transient species that occasionally pass through from other habitats (MacArthur and Wilson 1967) or facultatively utilize salt marsh habitats (Eggelston et al. 1998) as well as resident salt marsh species that facultatively utilize adjacent sub-habitats (Fukao 1980). Species spillover between sub-habitats and adjacent habitats (Tewfik and Bene 2003; Zeller et al. 2003) similarly can influence species occurrence. Hence, mainland salt marsh habitats

would be expected to contain a higher degree of habitat complexity and more species while small islands would be less complex and contain fewer species with large island salt marshes intermediate, as borne out by similarity analysis. Mainland salt marshes occupy one end of the estuarine spectrum, acting as refugia for marsh dependent nekton, while small island salt marshes act as refugia for open water nekton species that spill over into this habitat type from surrounding shallow water flats. Large island salt marshes provided a transitional intermediate habitat for both estuarine open water, or 'edge species', and salt marsh dependent nekton, 'interior species' (With and Crist 1995).

Additional evidence to support the idea that large island salt marshes function as an intermediate habitat was apparent from densities of salt marsh dependent and open water nekton species relative to mainland and small island salt marsh habitats. The density of salt marsh dependent species, including Fundulus heteroclitus, were an order of magnitude lower within the large island compared to the mainland salt marshes, and basically nonexistent within the small island salt marshes. These small salt marsh islands were apparently below the 'extinction threshold', for the minimum proportion of suitable habitat necessary for population persistence (Andren 1994; With and Crist 1995) for this species and other salt marsh dependents. The large salt marsh islands appeared to be of sufficient size and complexity to provide support for similar numbers of nekton species compared to mainland salt marshes, but the large island salt marshes apparently were not large enough to provide suitable habitats to support stable, independent populations of resident salt marsh nekton (Chapter 3). This density pattern for F. heteroclitus, related to salt marsh size, supports inferences on the linkage of population size to habitat patch size, and the existence of minimal size thresholds (Fahrig and Merriam 1985; Harrison et

al. 1988; Wauters et al. 1994; Dunning et al. 1995) predicted by the patch occupancy dynamic theory (Hanski 1994). Salt marsh dependent nekton usage patterns were similar to proposed mainland-island metapopulation models (Hanski 1994; Harrison and Taylor 1997). The contrary population size and density patterns of salt marsh dependent compared to open water nekton species, including Menidia menidia, submerged aquatic vegetation oriented Callinectes sapidus (Hovel and Lipcius 2002), which might utilize the marsh fringe (Peterson and Turner 1994), and the wider ranging competitor species Fundulus majalis (Abrams 1985; Weisburg 1986), suggests facultative salt marsh use by these species and further suggests that large island salt marshes were an intermediate within the examined estuarine salt marsh habitat spectrum.

Different patterns of estuarine habitat use by representative species, Fundulus heteroclitus and Lagodon rhomboides, were apparent from eel pot collections and supported observations from fyke and block net collections. It was apparent that F. heteroclitus exclusively utilized salt marsh habitats during high tide and not adjacent shallow water flats. During high tide, salt marshes, including island salt marshes, were preferentially use by F. heteroclitus over shallow water flats, possibly acting in similar fashion as coral patches serving as oases in reef lagoons for cryptic fishes (Alevizon et al. 1985) and seagrasses for fishes and shrimps (Fonseca et al. 1990). However, the bimodal distribution pattern observed for L. rhomboides, with high abundances observed within the low salt marsh and the interior of the adjacent shallow water flat, reflects the use of multiple habitats by L. rhomboides (Hettler 1989; Fonseca et al. 1990; 1996; Meyer et al. 1996). Salt marsh habitats are not essential for L. rhomboides populations as they are for F. heteroclitus populations and these habitats were only facultatively used. The

distribution pattern observed for L. rhomboides suggests that the expanse of shallow water estuarine flats might be more essential for population maintenance as a whole and low salt marsh represents a marginal habitat utilized only by smaller individuals. Hence, the high salt marsh habitat, especially of interior salt marshes, might act as low quality habitats for such a transient species.

It was evident that Fundulus heteroclitus preferred the shallow salt marsh areas of fringing mainland and island salt marshes over other habitats sampled and an apparent F. heteroclitus usage gradient is suspected to occur based on water depth (Ruiz et al. 1993). Fundulus heteroclitus populations are evidently not only restricted by a maximum water depth but this maximum water depth can restrict F. heteroclitus usage within salt marsh habitats. Ruiz et al. (1993) suggested that F. heteroclitus may be restricted to water depths of less than 1 m. While other factors might also influence distribution trends, including availability of prey (Weisberg and Lotrich 1986; Raichel et al. 2003) and interspecific competition (Baker-Dittus 1978; Weisburg 1986), the lack of a sufficient area deep enough to allow juvenile F. heteroclitus use yet shallow enough to restrict predator incursion into salt marsh habitat, is suspected to be a primary factor contributing to the consistent F. heteroclitus abundance patterns observed and follows the general size-water depth distribution for nekton suggested by Ruiz et al. (1993) and confirmed in this study.

The consistent size-water depth relationship observed for Fundulus heteroclitus and Lagodon rhomboides suggests a partitioning of the salt marsh habitat between different size classes for each species. The YOY F. heteroclitus utilize the shallow salt marsh region during seasonal periods when they are prevalent, and larger adult

individuals utilize the deeper salt marsh creek of the interior marshes and low salt marsh habitat regions. Similar size class specific partitioning of intertidal oyster reef habitat by xanthid crab species has been observed (Meyer 1994). These patterns initially appear contrary to observations by Kneib and Wagner (1994), who suggested that larger F. heteroclitus individuals ventured farther into the salt marshes they sampled (which were similar to my interior salt marshes), while smaller individuals occurred nearer the salt marsh/salt marsh creek interface. However, their results were consistent with my results examining catch for sampling locations located only within vegetated salt marsh. While size-water depth trends were not consistently apparent in interior salt marsh sites when only vegetated locations of the marshes were sampled, a larger scale fish size versus water depth pattern was obvious for F. heteroclitus, and L. rhomboides when all habitats within the interior salt marsh complex (including salt marsh creeks) were sampled. Kneib and Wagner (1994) would have missed such a pattern having not collected samples within the whole of the salt marsh complex they investigated.

The importance of shallow water predation refuge has been suggested as important for nekton population maintenance (Posey and Hines 1991; Ruiz et at. 1993). Water-depth dependent distribution patterns with significant use of the shallow salt marsh by Fundulus heteroclitus during high tides is similar to patterns observed by Kneib (1984) for YOY F. heteroclitus and Palaemonetes pugio in Georgia, USA. Observed relations of shallow salt marsh habitat size on resident nekton density patterns followed previous observations, including preferential usage of similar habitat by spotfin killifish (Fundulus luciae) (Able et al. 1983; Kneib 1984; Talbot and Able 1984; Able 1990), its suspected importance for F. heteroclitus population use (Able 1990) and YOY

distribution (Talbot and Able 1984), and importance for Lucania parva, (Able 1990) and Cyprinodon variegatus YOY occurrence (Talbot and Able 1984) and population maintenance (Able 1990; Chitty and Able 2004). The amount of shallow salt marsh habitat at the islands sampled during this study was identified as the leading factor, among numerous biotic, physical, environmental and habitat factors examined, for explaining resident salt marsh nekton, and particularly F. heteroclitus, density. Similarly, comparisons among salt marsh habitat based on size categories (mainland, large and small island) also indicated an increase in density for resident marsh nekton species including F. heteroclitus, F. luciae, C. variegatus and L. parva that proportionally followed the estimated shallow water salt marsh refuge area.

Previous investigators have suggested that utilization of low salt marsh was important in describing Fundulus heteroclitus population abundance (Kneib 1987; Kneib and Wagner 1994) along with the availability of suitable low tide habitat (Rozas and Odum 1987; McIvor and Odum 1988; Kneib and Wagner 1994). However, the amount of low salt marsh area surrounded by shallow water low tide refuge present at the marsh islands did not translate into equivalent increases in F. heteroclitus population abundance for the fringing salt marsh islands examined in this study. Instead, a direct relationship was observed between shallow salt marsh area and F. heteroclitus abundance. While seemingly contradictory to previous work, the types of salt marsh examined in each study must be considered. In this study the vertical range of the salt marshes sampled was almost 70 % of the 1.3 m tidal range for the region, with the salt marsh islands generally containing more extensive sloping low salt marsh area compared to high salt marsh flat. Previous work by Kneib and Wagner (1994) examined salt marsh creek habitat that

contained extensive areas of salt marsh flats at relatively high tidal elevations (within the upper 15 % of the 2.2 m tidal range) with high and low salt marsh sampling components separated by only 13 cm of vertical height, resulting in limited actual low salt marsh and relatively expansive high salt marsh habitat. In such salt marshes, predation risk on resident salt marsh species by high order predators would be reduced due to availability of extensive shallow salt marsh flats, and restricted access for predators via shallow water creeks. Hence, populations of F. heteroclitus might tend to be more abundant within creek fed salt marsh habitats that contain increased amounts of low salt marsh area adjacent to both shallow salt marsh and creek due to the combination of increased forage area and forage time allowed for F. heteroclitus populations within low salt marsh and predation refuge. By contrast, the present study examined island salt marsh habitats that were comprised of fringing salt marsh that contained no shallow water creeks but were surrounded by estuarine flats. Population size and density patterns of F. heteroclitus for fringing salt marshes, particularly isolated island salt marshes, might be more dependant on shallow salt marsh area due to the refuge this habitat provides for resident nekton larvae and juveniles (Talbot and Able 1984) and adults from predators. A similar hypothesis has been suggested based on previous work by Halpin (2000) that F. heteroclitus preferentially seek habitat that offers predation refuge over that with higher food resource and growth potential, intimating that reduction in predation risk is a major factor influencing F. heteroclitus distribution within salt marsh habitats. Based on F. heteroclitus abundance estimated from previous work (Kneib and Wagner 1994) and the current study, it appears that Shelford's law of tolerance might pertain to salt marsh dependant nekton populations. While both shallow and low salt marsh habitats have

been suggested to be important in the support and maintenance of resident salt marsh nekton populations, the habitat that is in the lesser amount might become a factor that can directly limit populations relative to minimal tolerable habitat size attributes, thus becoming critical habitat. This might be especially true in cases where the overall size of the habitats utilized by fauna and connectivity to like habitat is limited.

Because F. heteroclitus tend to utilize shallow water habitat, dispersal via movement corridors among salt marsh habitats might be restricted to shallow water shoals. Water depth that exceeds a maximum critical depth might restrict F. heteroclitus movement (Simberloff and Wilson 1969) and cause potential differences in immigration and emigration for F. heteroclitus between intertidal salt marsh habitats based not only on distance but also water depths separating like habitats. Seasonal opening and closing of shallow water movement corridors for resident salt marsh nekton, along shoals between salt marshes, is suspected to occur based on variations in tidal and predator conditions. The late fall and early spring time periods corresponded to annual periods of astronomically low tides compared to other times of the year (Hutchinson and Sklar 1993) and annual lows for predator concentrations within the shallow water habitats adjacent to salt marshes (Chapter 3). These time periods also correspond to the observed highest movement onto the shallow water flats by F. heteroclitus. Hence, late fall and early spring might be the times of year in which major movement of F. heteroclitus occurs along the postulated shallow water flat movement corridors.

Based on eel pot collections Fundulus heteroclitus populations at island, mainland and interior salt marsh sites were initially (mid spring) similar in terms of mean size. This initial similarity suggests that colonization of all salt marsh habitats might have

occurred by the previous years' cohort during the late fall to early spring (Chapter 4). Following this initial similarity in size composition for all salt marsh habitats, consistent increase in size disparity between the island salt marsh sites and those observed for the mainland and interior salt marsh sites suggests that the populations within the island salt marsh habitats are not self sustaining but are dependant on an influx of adult individuals from elsewhere. This lack of YOY contribution to the island salt marsh populations was particularly apparent during the late summer when YOY should have significantly contributed to the island salt marsh populations as they had at mainland and interior salt marsh populations (Chapter 3). Island salt marsh sites apparently acted as sinks for estuarine F. heteroclitus populations supplied from the mainlands and a mainland-island metapopulation model (Harrison and Taylor 1997) best explains F. heteroclitus utilization at the island salt marsh habitats.

Though many nekton species collected during this study did not show density distribution patterns relative to specific salt marsh habitat size, particularly species with good larval and adult dispersal ability, those species that have limited larval and adult dispersal ability, in particular Fundulus heteroclitus, did show consistent specific density patterns related to salt marsh size. It was apparent that the overall ecological function of the smaller isolated habitats were limited, and population support is habitat size-dependent for salt marsh resident species such as Fundulus heteroclitus. Along with population size and density pattern implications observed in the current study, nekton dispersal ability might significantly affect gene flow and sub-population mixing (Kelly et al. 2001; Fratini and Vannini 2002; Calsbeek and Smith 2003) for estuarine species. Transient species such as Lagodon rhomboides, which have the ability to move from salt

marsh to salt marsh via other estuarine habitats (Fonseca et al. 1990; 1996; Meyer et al. 1999; Hovell et al. 2002; Paperno et al. 2001), have substantial population mixing potential and probably create a single functional population. However, species with poor dispersal ability, such as Fundulus heteroclitus (Lotrich 1975; Abrams 1985), might be unable to consistently occupy island habitats and might form distinct segregated sub-populations similar to modeled metapopulations (Levins 1970; Harrison and Taylor 1997) as observed by Harrison et al. (1988) and Wahlberg et al. (2002) for the checkered butterfly, Euphydryas editha, which could be separated by geographic distance or temporal movement patterns. Based on my observations on the influence of salt marsh habitat size, including particular sub-habitats, and degree of isolation from like habitat on nekton utilization, it should be possible to not only predict nekton utilization for salt marsh habitats using these attributes in a predictive spatial model, but it should also be possible to specifically design salt marsh restorations to target utilization by specific nekton species. With such information, specific targeted goals for nekton use of managed and restored estuarine habitats is possible and should be attempted.

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Table 2-1. Mean annual physical parameters measured for the salt marsh types during fyke net (top) and eel pot (bottom) collections. Significant differences ($p \leq 0.05$) between marsh types are designated by different letters. Mainland salt marsh areas are minimal estimates based on the smallest mainland salt marsh site. Data that had to be nlog + 1 transformed prior to analysis to meet homogeneity assumptions are designated by a single asterisk, arcsine transformed data are designated by #. Data not meeting homogeneity assumptions after transformation and were analyzed using a non-parametric test are designated by two asterisks. Hyphens indicate that no observations were taken for that parameter at that site.

Fyke Net Collections:

| Marsh Type | Marsh Fringe Elevation (cm) | Distance Water Intrusion | Marsh Slope | Vertical Relief (cm) | Total Marsh* Size ($m^2$) | Low Marsh* Size ($m^2$) | Shallow* Marsh Size ($m^2$) | Distance** to Nearest Marsh (m) | Percent# Organic | Percent# Silt | Salinity (ppt) | Water Temp. ($^oC$) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | Year One | | | | | | |
| Mainland | -78.5 | 22.9 | 0.02753 | $45.2^{B}$ | $>76{,}000^{A}$ | $>19{,}300^{A}$ | $>56{,}700^{A}$ | $0^{B}$ | 6.03 | $28.35^{A}$ | 32.3 | 18.1 |
| Large Island | -81.5 | 25.8 | 0.02912 | $87.6^{A}$ | $5{,}605^{B}$ | $2{,}800^{B}$ | $2{,}805^{B}$ | $705^{A}$ | 2.23 | $21.96^{AB}$ | 32.0 | 19.0 |
| Small Island | -94.9 | 15.1 | 0.04008 | $73.1^{A}$ | $629^{C}$ | $582^{C}$ | $47^{C}$ | $439^{A}$ | 1.51 | $8.73^{B}$ | 33.7 | 17.9 |
| | | | | | | Year Two | | | | | | |
| Mainland | -68.4 | 20.6 | 0.03014 | $45.2^{B}$ | $>76{,}000^{A}$ | $>19{,}300^{A}$ | $>56{,}700^{A}$ | $0^{B}$ | 6.03 | $28.35^{A}$ | 32.4 | $17.6^{AB}$ |
| Large Island | -72.6 | 25.1 | 0.03283 | $87.6^{A}$ | $5{,}605^{B}$ | $2{,}800^{B}$ | $2{,}805^{B}$ | $705^{A}$ | 2.23 | $21.96^{AB}$ | 33.8 | $16.2^{B}$ |
| Small Island | -80.3 | 13.3 | 0.04969 | $73.1^{A}$ | $629^{C}$ | $582^{C}$ | $47^{C}$ | $439^{A}$ | 1.51 | $8.73^{B}$ | 31.3 | $19.7^{A}$ |

Eel Pot Collections:

| Marsh Type | Water depth (cm) | | | | | | | Salinity (ppt) | Water Temp. ($^oC$) |
|---|---|---|---|---|---|---|---|---|---|
| | Shallow Marsh | Marsh Fringe | 5 m | 25 m | 100 m | Midpoint | Marsh Creek | | |
| Interior | 15.6 | $26.5^{A}$ | --- | --- | --- | --- | 51.3 | 32.6 | 20.0 |
| Mainland | 17.8 | $38.8^{B}$ | 58.6 | 73.0 | 78.2 | 139.8 | --- | 32.7 | 19.1 |
| Island | 19.8 | $43.2^{B}$ | 60.7 | 74.0 | 85.8 | 139.8 | --- | 32.5 | 18.7 |

Table 2-2. Mean number of stems 1 $m^{-2}$ for each collection year. For each species during each collection year comparisons between mainland marsh (MLM), large island marsh (LIM) and small island marsh (SIM) treatments did not differ significantly ($p > 0.05$) from one another. Numbers in parentheses are one standard error, L = live stems, D = dead stems, T = total of all stems.

| | Year 1 | | | Year 2 | | |
|---|---|---|---|---|---|---|
| Species/Type | MLM | LIM | SIM | MLM | LIM | SIM |
| Spartina alterniflora (L) | 205.78(±23.99) | 139.89(±23.57) | 240.33(±41.91) | 233.94(±33.59) | 144.00(±17.13) | 239.22(±22.62) |
| Spartina alterniflora (D) | 186.22(±24.65) | 118.00(±3.83) | 142.11(±27.91) | 151.17(±24.34) | 101.78(±16.06) | 87.44(±12.09) |
| Spartina alterniflora (T) | 392.00(±46.56) | 257.89(±19.93) | 382.44(±59.71) | 385.11(±54.72) | 245.78(±33.10) | 326.67(±20.72) |
| Salicornia virginica (T) | 11.67(±7.62) | 0.00(±0.00) | 0.00(±0.00) | 1.78(±1.78) | 0.22(±0.22) | 0.00(±0.00) |
| Total stems all species | 403.89(±45.27) | 257.89(±19.93) | 382.44(±59.71) | 386.89(±55.67) | 246.00(±33.20) | 326.67(±20.72) |

Table 2-3. Number of individuals 10 $m^{-2}$ (cumulative for total years 1 and 2), for the fish and decapod species that comprised the top 95% of all individuals collected. For each species during each collection date, comparisons between mainland marsh (MLM), large island marsh (LIM) and small island marsh (SIM) treatments that are significantly different from one another at $p \leq 0.05$ are indicated by a different letter. Numbers in parentheses are one standard error.

| Date | June 2002 | | | November 2002 | | |
|---|---|---|---|---|---|---|
| Species | MLM | LIM | SIM | MLM | LIM | SIM |
| Cyprinodon variegatus | 32.30(±32.03) | 0.59(±0.39) | 0.00(±0.00) | 0.65(±0.55) | 0.03(±0.02) | 0.00(±0.00) |
| Eucinostomus spp. | 1.01(±0.96) | 0.02(±0.02) | 0.00(±0.00) | 8.92(±5.62) | 3.91(±3.82) | 0.42(±0.17) |
| Fundulus heteroclitus | 17.33(±7.99)$^{A}$ | 1.77(±0.90)$^{AB}$ | 0.17(±0.12)$^{B}$ | 6.91(±3.58)$^{A}$ | 0.30(±0.12)$^{AB}$ | 0.06(±0.06)$^{B}$ |
| Fundulus lucia | 0.04(±0.04) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus majalis | 2.34(±0.35) | 2.31(±0.34) | 8.76(±4.55) | 0.33(±0.15) | 0.46(±0.36) | 0.14(±0.14) |
| Gobionellus boleosoma | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.01(±0.01) | 0.00(±0.00) | 0.00(±0.00) |
| Gobiosoma bosc | 0.04(±0.03) | 0.00(±0.00) | 0.00(±0.00) | 0.35(±0.35) | 0.00(±0.00) | 0.00(±0.00) |
| Lagodon rhomboides | 14.77(±6.12) | 25.76(±7.72) | 15.52(±10.60) | 5.50(±1.79) | 5.56(±5.17) | 2.49(±1.07) |
| Leiostomus xanthurus | 1.12(±0.24) | 0.38(±0.32) | 0.91(±0.20) | 0.04(±0.02) | 0.14(±0.14) | 0.14(±0.14) |
| Lucania parva | 0.04(±0.04) | 0.06(±0.06) | 0.00(±0.00) | 13.59(±13.39) | 0.44(±0.44) | 0.00(±0.00) |
| Menidia menidia | 9.00(±5.48)$^{AB}$ | 0.00(±0.00)$^{B}$ | 70.57(±46.92)$^{A}$ | 1.81(±1.04) | 8.76(±8.52) | 21.05(±13.05) |
| Mugil cephalus | 0.05(±0.05) | 0.00(±0.00) | 0.00(±0.00) | 0.19(±0.17) | 0.05(±0.05) | 0.00(±0.00) |
| Mugil curema | 1.61(±0.54)$^{A}$ | 0.02(±0.02)$^{B}$ | 0.59(±0.24)$^{AB}$ | 0.04(±0.02) | 0.05(±0.05) | 0.00(±0.00) |
| Paralichthys albigutta | 0.08(±0.05)$^{B}$ | 0.31(±0.01)$^{AB}$ | 0.40(±0.08)$^{A}$ | 0.04(±0.02) | 0.11(±0.11) | 0.00(±0.00) |
| Paralichthys dentatus | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.01(±0.01) | 0.00(±0.00) | 0.00(±0.00) |
| Other Fishes | 0.16(±0.08) | 0.26(±0.14) | 0.17(±0.04) | 0.07(±0.06) | 0.09(±0.05) | 0.03(±0.03) |
| Callinectes sapidus | 2.28(±0.73) | 2.99(±0.43) | 1.86(±0.97) | 0.32(±0.14) | 0.12(±0.12) | 1.33(±0.90) |
| Callinectes similis | 0.05(±0.04) | 0.00(±0.00) | 2.06(±2.06) | 0.03(±0.03) | 0.00(±0.00) | 0.04(±0.04) |
| Clibanarius vittatus | 0.00(±0.00) | 0.02(±0.02) | 0.07(±0.07) | 0.13(±0.11) | 0.00(±0.00) | 0.18(±0.12) |
| Palaemonetes intermedius | 0.04(±0.03) | 0.18(±0.18) | 0.00(±0.00) | 1.08(±0.33) | 1.34(±1.34) | 1.55(±0.59) |
| Palaemonetes pugio | 0.73(±0.45) | 0.28(±0.22) | 0.00(±0.00) | 1.94(±0.82) | 0.19(±0.19) | 0.40(±0.40) |
| Palaemonetes vulgaris | 0.03(±0.03) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Farfantepenaeus aztecus | 0.59(±0.31) | 0.00(±0.00) | 0.18(±0.18) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Farfantepenaeus duorarum | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.14(±0.09) | 0.04(±0.02) | 0.24(±0.08) |
| Other Decapods | 0.03(±0.02) | 0.48(±0.36) | 0.03(±0.03) | 1.50(±1.15) | 0.05(±0.05) | 0.31(±0.11) |

Table 2-3. (Continued).

| Date | March 2003 | | | Total Year One | | |
|---|---|---|---|---|---|---|
| Species | MLM | LIM | SIM | MLM | LIM | SIM |
| Cyprinodon variegatus | 0.52(±0.31) | 0.06(±0.02) | 0.24(±0.08) | 33.47(±32.86) | 0.68(±0.41) | 0.24(±0.08) |
| Eucinostomus spp. | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 9.93(±5.44) | 3.93(±3.82) | 0.42(±0.17) |
| Fundulus heteroclitus | 4.10(±2.63) | 1.02(±0.72) | 0.06(±0.03) | 28.34(±11.02)$^{A}$ | 3. 09(±1.54)$^{B}$ | 0.29(±0.11)$^{B}$ |
| Fundulus lucia | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.04(±0.04) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus majalis | 1.08(±0.84) | 1.16(±1.10) | 15.93(±11.48) | 3.75(±1.02) | 3.93(±1.75) | 24.83(±15.72) |
| Gobionellus boleosoma | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.01(±0.01) | 0.00(±0.00) | 0.00(±0.00) |
| Gobiosoma bosc | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.39(±0.37) | 0.00(±0.00) | 0.00(±0.00) |
| Lagodon rhomboides | 19.94(±11.72) | 10.41(±8.46) | 2.35(±1.01) | 40.21(±14.89) | 41.73(±8.77) | 20.36(±11.98) |
| Leiostomus xanthurus | 49.73(±16.10) | 906.16(±878.33) | 51.55(±9.03) | 50.89(±6.12) | 906.68(±878.59) | 52.60(±9.34) |
| Lucania parva | 6.14(±6.00) | 0.53(±0.53) | 0.00(±0.00) | 19.77(±19.44) | 1.03(±0.53) | 0.00(±0.00) |
| Menidia menidia | 1.26(±1.09) | 5.54(±5.36) | 3.88(±2.90) | 12.07(±4.98) | 14.30(±7.35) | 95.50(±58.82) |
| Mugil cephalus | 4.99(±3.34)$^{A}$ | 0.28(±0.04)$^{B}$ | 0.00(±0.00)$^{C}$ | 5.23(±3.28)$^{A}$ | 0.33(±0.01)$^{B}$ | 0.00(±0.00)$^{C}$ |
| Mugil curema | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 1.65(±0.53) | 0.07(±0.05) | 0.59(±0.24) |
| Paralichthys albigutta | 0.00(±0.00) | 0.00(±0.00) | 0.03(±0.03) | 0.12(±0.05)$^{B}$ | 0.42(±0.10)$^{A}$ | 0.43(±0.07)$^{A}$ |
| Paralichthys dentatus | 0.69(±0.54) | 0.65(±0.53) | 0.43(±0.28) | 0.70(±0.54) | 0.65(±0.53) | 0.43(±0.28) |
| Other Fishes | 0.27(±0.08) | 0.08(±0.05) | 0.11(±0.07) | 0.50(±0.20) | 0.43(±0.14) | 0.31(±0.11) |
| Callinectes sapidus | 0.15(±0.08) | 0.08(±0.05) | 0.00(±0.00) | 2.75(±0.71) | 3.19(±0.48) | 3.19(±0.86) |
| Callinectes similis | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.08(±0.07) | 0.00(±0.00) | 2.10(±2.09) |
| Clibanarius vittatus | 0.00(±0.00) | 0.04(±0.04) | 0.00(±0.00) | 0.13(±0.11) | 0.06(±0.04) | 0.25(±0.13) |
| Palaemonetes intermedius | 17.15(±16.07) | 7.71(±0.75) | 0.00 (±0.00) | 18.27(±16.05) | 9.23(±6.80) | 1.55(±0.59) |
| Palaemonetes pugio | 3.65(±2.41) | 0.28(±0.21) | 0.03(±0.03) | 6.32(±2.70) | 0.75(±0.41) | 0.43(±0.39) |
| Palaemonetes vulgaris | 0.00(±0.00)$^{B}$ | 0.55(±0.28)$^{A}$ | 0.10(±0.10)$^{AB}$ | 0.03(±0.03)$^{B}$ | 0.55(±0.28)$^{A}$ | 0.10(±0.10)$^{B}$ |
| Farfantepenaeus aztecus | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.59(±0.31) | 0.00(±0.00) | 0.18(±0.18) |
| Farfantepenaeus duorarum | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.14(±0.09) | 0.04(±0.02) | 0.24(±0.08) |
| Other Decapods | 0.14(±0.07) | 0.31(±0.20) | 0.09(±0.09) | 1.67(±1.20) | 0.84(±0.45) | 0.43(±0.05) |

Table 2-3. (Continued).

| Date | June 2003 | | | November 2003 | | |
|---|---|---|---|---|---|---|
| Species | MLM | LIM | SIM | MLM | LIM | SIM |
| Cyprinodon variegatus | 0.41(±0.20) | 0.06(±0.06) | 0.00(±0.00) | 7.47(±4.27) | 10.73(±5.80) | 0.00(±0.00) |
| Eucinostomus spp. | 0.06(±0.06) | 0.00(±0.00) | 0.00(±0.00) | 0.43(±0.31) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus heteroclitus | 7.19(±3.60)$^{A}$ | 0.59(±0.30)$^{B}$ | 0.35(±0.24)$^{B}$ | 11.64(±8.62)$^{A}$ | 1.89(±1.32)$^{A}$ | 0.00(±0.00)$^{B}$ |
| Fundulus lucia | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus majalis | 0.74(±0.38) | 0.08(±0.05) | 40.60(±39.23) | 2.77(±1.26) | 3.11(±2.09) | 1.87(±1.82) |
| Gobionellus boleosoma | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Gobiosoma bosc | 0.12(±0.12) | 0.00(±0.00) | 0.00(±0.00) | 0.99(±0.87) | 0.02(±0.02) | 0.00(±0.00) |
| Lagodon rhomboides | 17.63(±8.61) | 15.61(±0.75) | 36.23(±16.17) | 3.72(±2.10) | 0.23(±0.14) | 6.15(±3.76) |
| Leiostomus xanthurus | 2.75(±0.46) | 1.67(±0.93) | 3.32(±0.89) | 0.60(±0.42) | 0.00(±0.00) | 0.72(±0.72) |
| Lucania parva | 0.03(±0.03) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) |
| Menidia menidia | 1.35(±1.14) | 0.24(±0.16) | 30.99(±29.72) | 0.30(±0.12) | 0.60(±0.30) | 0.78(±0.58) |
| Mugil cephalus | 0.67(±0.63) | 0.52(±0.52) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Mugil curema | 0.06(±0.06) | 0.00(±0.00) | 0.18(±0.09) | 0.01(±0.01) | 0.00(±0.00) | 0.00(±0.00) |
| Paralichthys albigutta | 0.03(±0.03) | 0.24(±0.14) | 0.11(±0.07) | 0.05(±0.05) | 0.04(±0.04) | 0.08(±0.08) |
| Paralichthys dentatus | 0.12(±0.05) | 0.14(±0.04) | 0.35(±0.30) | 0.04(±0.04) | 0.00(±0.00) | 0.06(±0.03) |
| Other Fishes | 0.06(±0.04) | 0.95(±0.71) | 0.82(±0.82) | 0.11(±0.08) | 0.02(±0.02) | 0.13(±0.02) |
| Callinectes sapidus | 4.88(±1.32) | 3.05(±0.47) | 5.95(±1.10) | 0.46(±0.22)$^{B}$ | 0.21(±0.11)$^{B}$ | 1.81(±0.23)$^{A}$ |
| Callinectes similis | 0.00(±0.00) | 0.04(±0.04) | 0.38(±0.38) | 0.00(±0.00) | 0.00(±0.00) | 0.20(±0.20) |
| Clibanarius vittatus | 2.74(±2.62) | 2.91(±1.46) | 2.79(±1.66) | 0.03(±0.03) | 0.00(±0.00) | 0.36(±0.36) |
| Palaemonetes intermedius | 0.00(±0.00) | 0.18(±0.10) | 0.06(±0.06) | 0.03(±0.02) | 0.04(±0.04) | 0.00(±0.00) |
| Palaemonetes pugio | 1.21(±0.87) | 0.00(±0.00) | 0.17(±0.07) | 0.11(±0.09) | 1.42(±1.36) | 0.00(±0.00) |
| Palaemonetes vulgaris | 0.01(±0.01) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.06(±0.06) | 0.00(±0.00) |
| Farfantepenaeus aztecus | 1.16(±0.73) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Farfantepenaeus duorarum | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.11(±0.08) | 0.02(±0.02) | 0.11(±0.07) |
| Other Decapods | 0.03(±0.03)$^{B}$ | 0.16(±0.05)$^{A}$ | 0.00(±0.00)$^{B}$ | 0.45(±0.42) | 0.04(±0.04) | 0.31(±0.26) |

Table 2-3. (Continued).

| Date | March 2004 | | | Total Year Two | | |
|---|---|---|---|---|---|---|
| Species | MLM | LIM | SIM | MLM | LIM | SIM |
| Cyprinodon variegatus | 0.94(±0.46)$^{A}$ | 0.03(±0.03)$^{B}$ | 0.16(±0.05)$^{AB}$ | 8.82(±4.68) | 10.82(±5.82) | 0.16(±0.05) |
| Eucinostomus spp. | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.49(±0.30) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus heteroclitus | 1.74(±0.77)$^{A}$ | 0.05(±0.03)$^{B}$ | 0.51(±0.46)$^{AB}$ | 20.57(±9.04)$^{A}$ | 2.53(±1.35)$^{B}$ | 0.86(±0.70)$^{B}$ |
| Fundulus lucia | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus majalis | 1.68(±1.00) | 1.48(±0.91) | 2.71(±0.78) | 5.19(±1.75) | 4.67(±2.96) | 45.18(±38.30) |
| Gobionellus boleosoma | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Gobiosoma bosc | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 1.11(±0.98) | 0.02(±0.02) | 0.00(±0.00) |
| Lagodon rhomboides | 5.94(±1.82) | 20.94(±16.74) | 8.48(±7.74) | 27.29(±10.85) | 36.78(±17.61) | 50.86(±25.09) |
| Leiostomus xanthurus | 58.30(±14.12)$^{AB}$ | 122.50(±66.80)$^{A}$ | 11.66(±2.10)$^{B}$ | 61.65(±14.13) | 124.17(±66.70) | 15.70(±2.33) |
| Lucania parva | 5.14(±5.13) | 0.00(±0.00) | 0.00(±0.00) | 5.17(±5.12) | 0.02(±0.02) | 0.00(±0.00) |
| Menidia menidia | 1.17(±0.88) | 0.44(±0.09) | 0.27(±0.27) | 2.82(±1.20) | 1.28(±0.42) | 32.04(±29.59) |
| Mugil cephalus | 1.81(±1.20)$^{A}$ | 0.62(±0.04)$^{A}$ | 0.00(±0.00)$^{B}$ | 2.48(±1.82)$^{A}$ | 1.14(±0.55)$^{A}$ | 0.00(±0.00)$^{B}$ |
| Mugil curema | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.07(±0.06) | 0.00(±0.00) | 0.18(±0.09) |
| Paralichthys albigutta | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.08(±0.05) | 0.28(±0.15) | 0.19(±0.15) |
| Paralichthys dentatus | 0.68(±0.57) | 0.66(±0.23) | 0.42(±0.05) | 0.84(±0.64) | 0.80(±0.26) | 0.83(±0.24) |
| Other Fishes | 0.01(±0.01) | 0.02(±0.02) | 0.07(±0.04) | 0.18(±0.07) | 0.99(±0.72) | 1.02(±0.82) |
| Callinectes sapidus | 0.08(±0.03)$^{AB}$ | 0.35(±0.15)$^{A}$ | 0.00(±0.00)$^{B}$ | 5.42(±1.32) | 3.61(±0.59) | 7.76(±1.25) |
| Callinectes similis | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00)$^{B}$ | 0.04(±0.04)$^{AB}$ | 0.58(±0.33)$^{A}$ |
| Clibanarius vittatus | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 2.77(±2.66) | 2.91(±1.46) | 3.15(±1.99) |
| Palaemonetes intermedius | 1.40(±1.26) | 0.71(±0.24) | 0.14(±0.14) | 1.43(±1.26) | 0.93(±0.17) | 0.20(±0.12) |
| Palaemonetes pugio | 6.93(±4.00)$^{A}$ | 1.80(±0.87)$^{A}$ | 0.03(±0.03)$^{B}$ | 8.25(±4.37)$^{A}$ | 3.22(±2.15)$^{A}$ | 0.20(±0.11)$^{B}$ |
| Palaemonetes vulgaris | 0.01(±0.01)$^{B}$ | 2.06(±0.93)$^{A}$ | 0.00(±0.00)$^{B}$ | 0.02(±0.01)$^{B}$ | 2.12(±0.98)$^{A}$ | 0.00(±0.00)$^{B}$ |
| Farfantepenaeus aztecus | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) | 1.16(±0.73) | 0.02(±0.02) | 0.00(±0.00) |
| Farfantepenaeus duorarum | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.11(±0.08) | 0.02(±0.02) | 0.11(±0.07) |
| Other Fishes | 0.30(±0.18) | 0.13(±0.05) | 0.47(±0.43) | 0.78(±0.44) | 0.33(±0.03) | 0.78(±0.39) |

Table 2-4. Principal component analysis for mean physical and biological parameters measured, and for cumulative nekton abundance for years one and two combined for the island salt marshes. Eigenvector eigenvalues with the highest loading for individual attributes are indicated by an asterisk.

| Attribute | Principal Component 1 | Principal Component 2 | Principal Component 3 | Principal Component 4 |
|---|---|---|---|---|
| Nekton Abundance | 0.0793 | -0.2667 | -0.4441* | 0.4478* |
| Salinity | 0.0547 | -0.0310 | 0.5904* | 0.4408* |
| Water Temperature | -0.3503* | 0.0294 | 0.1445 | -0.3130 |
| Total Marsh | 0.3564* | 0.1366 | 0.2336 | 0.0362 |
| Shallow Marsh Size | 0.3252* | 0.2818 | -0.2193 | 0.1607 |
| Low Marsh Size | 0.3552* | -0.0188 | 0.2928 | -0.0587 |
| Vertical Relief | 0.3069* | 0.0152 | 0.3545 | -0.0927 |
| Stem Density | -0.2838 | 0.3296 | -0.0399 | 0.2180 |
| Canopy Height | 0.2500 | -0.1286 | -0.1605 | -0.5910* |
| Stem Diameter | 0.3135* | -0.3748 | -0.1217 | -0.0553 |
| Nearest Salt Marsh | 0.2815 | -0.1944 | -0.1677 | 0.2394 |
| Silt-Clay Content | 0.2793 | 0.4204* | -0.1189 | -0.1018 |
| Organic Content | 0.1276 | 0.5901* | -0.1866 | 0.0222 |
| | --- | --- | --- | --- |
| Eigenvalue | 6.4177 | 2.3783 | 1.9062 | 1.4615 |
| Proportion | 0.4937 | 0.1829 | 0.1466 | 0.1124 |

Table 2-5. Stepwise multiple linear regression loaded with habitat attributes and $r^2$ contribution to assess island attribute influence to site cumulative abundance for individual species and functional groups. Attribute entry level into the model equaled $p < 0.15$. Signs in parentheses represent regression nature. * indicates that data for that species, group or attribute were nlog +1 transformed and # indicates arcsine transformation to conform data to assumptions of data homogeneity.

| Species/Group | Total* Marsh | Shallow* Marsh | Stem Density | Canopy Height | Silt# Content | Organic# Content | Mean Salinity | Mean Water Temperature | Vertical Range | Nearest Marsh | Model $r^2$ | p |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Callinectes sapidus* | 0.9079(-) | — | — | — | — | — | 0.0092(-) | — | 0.0012(+) | 0.0817(-) | 1.0000 | 0.0024 |
| Callinectes similis* | — | 0.5623(-) | — | — | — | — | 0.3219(+) | — | — | — | 0.8892 | 0.0394 |
| Clibanarius vittatus* | — | — | — | — | — | — | — | — | — | — | — | — |
| Cyprinodon variegatus* | 0.5103(+) | — | — | 0.1518(-) | — | — | — | 0.0070(-) | 0.3307(-) | — | 0.9999 | 0.0016 |
| Eucinostomus spp.* | — | — | — | — | — | — | — | — | — | — | — | — |
| Fundulus heteroclitus* | — | 0.8916(+) | — | — | — | — | — | — | — | — | 0.8916 | 0.0046 |
| Fundulus majalis* | — | 0.7935(-) | — | — | — | — | — | — | — | — | 0.7935 | 0.0172 |
| Lagodon rhomboides* | — | 0.1936(+ ) | — | — | — | — | 0.6903(-) | — | — | — | 0.8839 | 0.0396 |
| Leiostomus xanthurus* | — | — | — | — | — | — | — | — | — | — | — | — |
| Lucania parva* | — | 0.4975(+) | 0.0427(-) | — | — | — | 0.2899(+) | — | 0.1688(-) | — | 0.9989 | 0.0496 |
| Menidia menidia* | — | — | — | — | — | 0.1467(+) | 0.3601(-) | 0.4901(+) | 0.0031(-) | — | 1.0000 | 0.0058 |
| Mugil cephalus* | 0.9018(+) | — | — | — | 0.0776(+) | — | — | — | 0.0179(-) | — | 0.9973 | 0.0041 |
| Mugil curema* | — | — | — | — | — | — | — | — | — | — | — | — |
| Palaemonetes intermedius* | — | — | — | — | — | — | — | — | — | — | — | — |
| Palaemonetes pugio* | — | — | — | — | 0.7455(+) | — | — | — | — | — | 0.7455 | 0.0267 |
| Palaemonetes vulgaris* | — | — | — | — | — | — | — | 0.7801(-) | — | — | 0.7801 | 0.0197 |
| Farfantepenaeus aztecus* | — | — | — | — | — | — | — | — | — | — | — | — |
| Farfantepenaeus duorarum* | — | — | — | — | — | — | — | 0.8712(+) | 0.0787(-) | — | 0.9499 | 0.0065 |
| Paralichthys albigutta* | — | — | — | — | — | — | — | — | — | — | — | — |
| Paralichthys dentatus* | — | — | — | — | — | — | — | — | — | — | — | — |
| Resident Nekton* | — | 0.7760(+) | — | — | — | — | — | — | — | — | 0.7760 | 0.0204 |
| Transient Nekton* | — | — | — | — | — | — | — | — | — | — | — | — |
| All Fish* | — | — | — | — | — | — | — | — | — | — | — | — |
| All Decapods* | — | — | — | — | — | — | — | — | — | — | — | — |
| All Nekton* | — | — | — | — | — | — | — | — | — | — | — | — |

Table 2-6. Results of one factor Analysis of Similarities and Similarity Percentages analyses conducted on nekton species collected during years one and two fyke and block net collections, by collection season. All data for comparisons between salt marsh types were square root transformed. Similarity ranking values, R, (0.0 = most similar, 1.0 = least similar) and average percent dissimilarity were calculated for salt marsh type comparisons during each collection season. Significance level $p \leq 0.05$,

| Treatment Comparisons | Month | R Statistic | Significance Level | Average Percent Dissimilarity |
|---|---|---|---|---|
| Mainland/Large Island | June | 0.245 | 0.018 | 47.47 |
| Large Island/Small Island | June | 0.463 | 0.002 | 54.29 |
| Mainland/Small Island | June | 0.466 | 0.002 | 56.75 |
| Mainland/Large Island | November | 0.124 | 0.108 | 69.38 |
| Large Island/Small Island | November | 0.354 | 0.022 | 73.83 |
| Mainland/Small Island | November | 0.238 | 0.035 | 60.21 |
| Mainland/Large Island | March | 0.217 | 0.054 | 53.11 |
| Large Island/Small Island | March | 0.326 | 0.004 | 55.99 |
| Mainland/Small Island | March | 0.204 | 0.049 | 49.59 |

Table 2-7. Nekton species that contributed ten percent or greater toward discriminating between salt marsh type community composition in Analysis of Similarities and Similarity Percentages analyses.

| Species | Treatment Comparison | Month | Percent Contribution |
|---|---|---|---|
| Fundulus heteroclitus | Mainland/Large Island | June | 14.86 |
| Lagodon rhomboides | Mainland/Large Island | June | 13.88 |
| | | | |
| Menidia menidia | Large Island/Small Island | June | 28.02 |
| Fundulus majalis | Large Island/Small Island | June | 16.82 |
| Lagodon rhomboides | Large Island/Small Island | June | 14.48 |
| | | | |
| Menidia menidia | Mainland/Small Island | June | 22.74 |
| Fundulus majalis | Mainland/Small Island | June | 13.75 |
| Fundulus heteroclitus | Mainland/Small Island | June | 12.92 |
| Lagodon rhomboides | Mainland/Small Island | June | 12.27 |
| | | | |
| Fundulus heteroclitus | Mainland/Large Island | November | 15.30 |
| Cyprinodon variegatus | Mainland/Large Island | November | 13.34 |
| Lagodon rhomboides | Mainland/Large Island | November | 12.26 |
| | | | |
| Menidia menidia | Large Island/Small Island | November | 14.95 |
| Fundulus heteroclitus | Large Island/Small Island | November | 12.46 |
| Cyprinodon variegatus | Large Island/Small Island | November | 11.96 |
| Lagodon rhomboides | Large Island/Small Island | November | 11.69 |
| | | | |
| Menidia menidia | Mainland/Small Island | November | 15.61 |
| Lagodon rhomboides | Mainland/Small Island | November | 10.48 |
| | | | |
| Leiostomus xanthurus | Mainland/Large Island | March | 36.35 |
| Lagodon rhomboides | Mainland/Large Island | March | 11.41 |
| | | | |
| Leiostomus xanthurus | Large Island/Small Island | March | 39.74 |
| Fundulus majalis | Large Island/Small Island | March | 10.84 |
| Lagodon rhomboides | Large Island/Small Island | March | 10.62 |
| | | | |
| Leiostomus xanthurus | Mainland/Small Island | March | 19.01 |
| Lagodon rhomboides | Mainland/Small Island | March | 13.52 |
| Fundulus majalis | Mainland/Small Island | March | 11.43 |

Table 2-8. Mean number of individuals collected per hour with eel pots based on salt marsh and shallow water flat location, by salt marsh type and date. Location mean comparisons within a salt marsh type, for each date that are significantly different ($p \leq 0.05$) from one another are indicated by a different letter. Hyphens indicate that no samples were collected for that salt marsh type for that location. One standard error is indicated in parentheses. Catch for both species during 1/04 and 3/04 was zero or so low that reasonable comparisons could not be made, hence data were not included in this table.

| | | | Sample Location | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Marsh Type | Date | Species | Shallow | -2 m | 0 m | 5 m | 25 m | 100 m | Midpoint | Marsh Creek |
| Interior | 11/03 | F. heteroclitus | 68.6(0.5) | 112.6(34.8) | 170.7(76.2) | --- | --- | --- | --- | 92.1(73.0) |
| Interior | 11/03 | L. rhomboides | 0.0(0.0) | 0.1(0.1) | 0.7(0.7) | --- | --- | --- | --- | 4.9(2.9) |
| Interior | 5/04 | F. heteroclitus | 13.5(8.8) | 28.8(28.8) | 80.9(30.6) | --- | --- | --- | --- | 43.8(27.1) |
| Interior | 5/04 | L. rhomboides | 0.0(0.0) | 2.3(1.7) | 1.0(1.0) | --- | --- | --- | --- | 5.9(3.8) |
| Interior | 7/04 | F. heteroclitus | 42.7(14.2) | 51.6(3.6) | 38.8(12.3) | --- | --- | --- | --- | 8.5(4.9) |
| Interior | 7/04 | L. rhomboides | 3.3(3.3) | 3.0(1.6) | 25.8(16.6) | --- | --- | --- | --- | 9.0(3.9) |
| Interior | 9/04 | F. heteroclitus | $^{A}$127.2(37.1) | $^{B}$13.7(7.5) | $^{B}$5.6(3.5) | --- | --- | --- | --- | $^{B}$0.0(0.0) |
| Interior | 9/04 | L. rhomboides | 3.7(3.0) | 14.1(8.3) | 13.4(5.7) | --- | --- | --- | --- | 21.2(13.5) |
| Mainland | 11/03 | F. heteroclitus | $^{A}$87.4(25.3) | $^{B}$21.5(19.3) | $^{BC}$9.9(6.3) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | --- |
| Mainland | 11/03 | L. rhomboides | 0.0(0.0) | 1.2(0.8) | 2.7(2.1) | 3.7(2.8) | 6.5(3.1) | 6.1(4.9) | 0.6(0.5) | --- |
| Mainland | 5/04 | F. heteroclitus | $^{A}$16.7(14.5) | $^{A}$9.2(3.9) | $^{B}$1.1(0.5) | $^{B}$0.3(0.3) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | --- |
| Mainland | 5/04 | L. rhomboides | 0.7(0.7) | 16.6(10.4) | 22.9(5.8) | 22.0(15.0) | 27.1(12.0) | 24.4(10.7) | 0.8(0.4) | --- |
| Mainland | 7/04 | F. heteroclitus | $^{A}$32.2(11.9) | $^{B}$8.6(4.8) | $^{C}$2.5(2.3) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | --- |
| Mainland | 7/04 | L. rhomboides | 33.2(29.1) | 69.0(33.8) | 32.8(11.1) | 11.2(4.0) | 44.6(20.8) | 72.9(35.5) | 1.3(0.9) | --- |
| Mainland | 9/04 | F. heteroclitus | $^{A}$70.0(28.8) | $^{B}$0.6(0.3) | $^{C}$0.3(0.3) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | $^{C}$0.0(0.0) | --- |
| Mainland | 9/04 | L. rhomboides | $^{AB}$12.3(4.8) | $^{A}$35.3(11.6) | $^{A}$22.1(7.0) | $^{AB}$4.9(3.0) | $^{AB}$18.7(6.8) | $^{AB}$14.4(11.0) | $^{B}$0.4(0.4) | --- |
| Island | 11/03 | F. heteroclitus | $^{A}$5.0(3.6) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | --- |
| Island | 11/03 | L. rhomboides | 0.1(0.1) | 0.2(0.2) | 1.1(0.5) | 1.3(1.2) | 0.8(0.3) | 0.2(0.1) | 0.6(0.5) | --- |
| Island | 5/04 | F. heteroclitus | 1.0(1.0) | 1.4((1.1) | 0.0(0.0) | 0.0(0.0) | 0.0(0.0) | 0.0(0.0) | 0.0(0.0) | --- |
| Island | 5/04 | L. rhomboides | 0.3(0.3) | 7.0(4.2) | 7.7(4.4) | 3.6(3.6) | 6.1(5.9) | 11.2(6.4) | 0.8(0.3) | --- |
| Island | 7/04 | F. heteroclitus | $^{A}$15.5(12.2) | $^{AB}$0.4(0.2) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | --- |
| Island | 7/04 | L. rhomboides | $^{AB}$25.5(10.1) | $^{A}$43.6(12.0) | $^{AB}$25.4(9.9) | $^{AB}$12.8(12.5) | $^{AB}$14.4(13.0) | $^{AB}$58.3(35.0) | $^{B}$1.3(0.9) | --- |
| Island | 9/04 | F. heteroclitus | $^{A}$3.1(1.5) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | $^{B}$0.0(0.0) | --- |
| Island | 9/04 | L. rhomboides | $^{AB}$13.6(7.3) | $^{A}$25.7(9.2) | $^{AB}$24.3(8.2) | $^{B}$3.2(3.1) | $^{B}$2.8(2.8) | $^{AB}$22.7(11.4) | $^{B}$0.4(0.4) | --- |

Table 2-9. Linear regression results (natural log transformed data) examining average water depth versus catch per hour for Fundulus heteroclitus (FUH)and Lagodon rhomboides (LAR) from high tide eel pot collections for the various salt marsh types sampled. Few or no individuals of both target species were collected during January and March 2004 so analyses for these dates are not included in this table. NS = no significant difference at the $p \leq 0.05$ level.

| | | Water Depth (all traps) | | | | | Water Depth (marsh traps only) | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Date | Intercept | slope | $r^2$ | p | n | Intercept | slope | $r^2$ | p | n |
| | | | | | Interior Marshes | | | | | | |
| FUH | 11/03 | NS | NS | NS | NS | 11 | NS | NS | NS | NS | 8 |
| FUH | 5/04 | NS | NS | NS | NS | 10 | -0.23 | 0.200 | 0.86 | 0.0026 | 7 |
| FUH | 7/04 | 5.02 | -0.053 | 0.46 | 0.0147 | 12 | NS | NS | NS | NS | 9 |
| FUH | 9/04 | 6.05 | -0.100 | 0.73 | 0.0004 | 12 | 7.16 | -0.134 | 0.66 | 0.0082 | 9 |
| LAR | 11/03 | -0.38 | 0.036 | 0.40 | 0.0382 | 11 | NS | NS | NS | NS | 8 |
| LAR | 5/04 | NS | NS | NS | NS | 10 | NS | NS | NS | NS | 7 |
| LAR | 7/04 | NS | NS | NS | NS | 12 | NS | NS | NS | NS | 9 |
| LAR | 9/04 | NS | NS | NS | NS | 12 | NS | NS | NS | NS | 9 |
| | | | | | Mainland Marshes | | | | | | |
| FUH | 11/03 | 2.3 | -0.024 | 0.32 | <0.0001 | 42 | 4.82 | -0.107 | 0.57 | 0.0003 | 18 |
| FUH | 5/04 | 1.2 | -0.010 | 0.22 | 0.0026 | 40 | NS | NS | NS | NS | 16 |
| FUH | 7/04 | 1.8 | -0.015 | 0.28 | 0.0004 | 41 | 5.09 | -0.089 | 0.57 | 0.0005 | 17 |
| FUH | 9/04 | 1.8 | -0.014 | 0.24 | 0.0009 | 42 | 5.28 | -0.083 | 0.58 | 0.0002 | 18 |
| LAR | 11/03 | NS | NS | NS | NS | 42 | NS | NS | NS | NS | 18 |
| LAR | 5/04 | NS | NS | NS | NS | 40 | -1.09 | 0.104 | 0.63 | 0.0003 | 16 |
| LAR | 7/04 | 3.6 | -0.016 | 0.17 | 0.0076 | 41 | NS | NS | NS | NS | 17 |
| LAR | 9/04 | 3.0 | -0.013 | 0.17 | 0.0065 | 42 | NS | NS | NS | NS | 18 |
| | | | | | Island Marshes | | | | | | |
| FUH | 11/03 | NS | NS | NS | NS | 41 | NS | NS | NS | NS | 18 |
| FUH | 5/04 | 0.33 | -0.003 | 0.10 | 0.05 | 40 | NS | NS | NS | NS | 17 |
| FUH | 7/04 | 0.65 | -0.006 | 0.10 | 0.043 | 40 | NS | NS | NS | NS | 17 |
| FUH | 9/04 | 0.15 | -0.003 | 0.11 | 0.0359 | 41 | 1.26 | -0.019 | 0.25 | 0.03339 | 18 |
| LAR | 11/03 | NS | NS | NS | NS | 41 | -0.08 | 0.014 | 0.22 | 0.048 | 18 |
| LAR | 5/04 | NS | NS | NS | NS | 40 | NS | NS | NS | NS | 17 |
| LAR | 7/04 | 3.36 | -0.018 | 0.21 | 0.003 | 40 | NS | NS | NS | NS | 17 |
| LAR | 9/04 | 2.49 | -0.012 | 0.13 | 0.0213 | 41 | NS | NS | NS | NS | 18 |
| | | | | | All Marshes | | | | | | |
| FUH | 11/03 | 2.10 | -0.021 | 0.22 | <0.0001 | 94 | 3.63 | -0.076 | 0.23 | 0.0009 | 44 |
| FUH | 5/04 | 1.38 | -0.012 | 0.17 | <0.0001 | 90 | 2.33 | -0.040 | 0.15 | 0.0122 | 40 |
| FUH | 7/04 | 1.94 | -0.016 | 0.25 | <0.0001 | 93 | 4.42 | -0.074 | 0.42 | <0.0001 | 43 |
| FUH | 9/04 | 1.60 | -0.013 | 0.21 | <0.0001 | 95 | 4.04 | -0.063 | 0.42 | <0.0001 | 45 |
| LAR | 11/03 | NS | NS | NS | NS | 94 | -0.10 | 0.018 | 0.17 | 0.0055 | 44 |
| LAR | 5/04 | NS | NS | NS | NS | 90 | -0.11 | 0.046 | 0.23 | 0.0014 | 40 |
| LAR | 7/04 | 3.07 | -0.014 | 0.12 | 0.0008 | 93 | NS | NS | NS | NS | 43 |
| LAR | 9/04 | 2.62 | -0.011 | 0.12 | 0.0005 | 95 | NS | NS | NS | NS | 45 |

Table 2-10. Mean size (mm standard length) of individuals collected with eel pots based on salt marsh and shallow water flat location, by salt marsh type and date. Location mean comparisons among salt marsh types, for each date that are significantly different ($p \leq 0.05$) from one another are indicated by a different letter. Hyphens indicate that no samples were collected for that salt marsh type for that location. One standard error is indicated in parentheses. Catch for both species during 1/04 and 3/04 was zero or so low that reasonable comparisons could not be made hence data were not included in this table.

| Date | Species | Sample Location | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Shallow | | | Low | | | 5 m | | 25 m | | 100 m | |
| | | Interior | Mainland | Island | Interior | Mainland | Island | Mainland | Island | Mainland | Island | Mainland | Island |
| 11/03 | F. heteroclitus | $^{C}$37.9(0.6) | $^{B}$40.7(0.6) | $^{A}$47.8(1.7) | 44.2(0.7) | 44.8(0.8) | --- | --- | --- | --- | --- | --- | --- |
| 11/03 | L. rhomboides | --- | --- | --- | 49.6(2.6) | 50.5(1.3) | 53.1(1.7) | 54.0(1.6) | 57.9(2.6) | 52.0(1.0) | 55.2(2.8) | $^{B}$48.4(1.2) | $^{A}$78.7(13.8) |
| 5/04 | F. heteroclitus | 48.9(1.7) | 50.1(0.9) | 44.2(2.1) | $^{B}$46.0(0.6) | $^{A}$52.3(0.8) | $^{A}$55.2(2.8) | --- | --- | --- | --- | --- | --- |
| 5/04 | L. rhomboides | --- | --- | --- | 29.0(1.0) | 28.6(0.4) | 28.6(0.4) | $^{B}$26.9(0.5) | $^{A}$30.3(0.7) | $^{B}$29.0(0.7) | $^{A}$32.0(0.8) | $^{B}$30.5(0.6) | $^{A}$32.1(0.5) |
| 7/04 | F. heteroclitus | $^{B}$44.2(1.1) | $^{B}$46.6(0.9) | $^{A}$55.3(1.9) | $^{B}$47.6(1.0) | $^{AB}$52.2(1.3) | $^{A}$57.0(11.4) | --- | --- | --- | --- | --- | --- |
| 7/04 | L. rhomboides | $^{B}$37.6(1.0) | $^{B}$38.5(0.8) | $^{A}$46.1(0.7) | $^{B}$44.2(0.9) | $^{AB}$45.1(0.6) | $^{A}$47.0(0.5) | $^{A}$50.0(0.6) | $^{B}$42.1(1.1) | 47.9(0.6) | 47.6(1.2) | $^{B}$44.8(0.8 ) | $^{A}$50.7(0.8) |
| 9/04 | F. heteroclitus | $^{B}$40.4(1.1) | $^{B}$38.2(0.7) | $^{A}$55.6(2.4) | $^{B}$40.0(1.2) | $^{A}$59.8(4.6) | --- | --- | --- | --- | --- | --- | --- |
| 9/04 | L. rhomboides | $^{B}$45.6(2.2) | $^{B}$47.5(1.0) | $^{A}$59.4(1.0) | 57.0(1.1) | 56.0(0.7) | 57.6(0.6) | 62.7(1.0) | 59.8(1.9) | $^{B}$60.6(1.0) | $^{A}$66.9(1.5) | 64.2(2.5) | 64.6(1.5) |

Table 2-11. Comparison of Fundulus heteroclitus and Lagodon rhomboides mean presence (1) and absences (0) at set positions from salt marsh during ebb tide eel pot collections. For date comparisons of the occurrence for each species for each salt marsh type, mainland salt marsh (mainland), island salt marsh (island) and interior salt marsh (interior), those dates that are significantly different from one another are indicated by a different letter ($p \leq 0.05$).

| Species | Marsh Type | Location | Date | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | November 2003 | January 2004 | March 2004 | May 2002 | July 2004 | September 2004 |
| Fundulus heteroclitus | Mainland | 25 m | $0.7^{A}$ | $0.2^{AB}$ | $0.7^{A}$ | $0.0^{B}$ | $0.3^{AB}$ | $0.0^{B}$ |
| Fundulus heteroclitus | Island | 25 m | 0.3 | 0.0 | 0.2 | 0.0 | 0.0 | 0.0 |
| Lagodon rhomboides | Mainland | 25 m | $0.8^{A}$ | $0.0^{B}$ | $0.0^{B}$ | $1.0^{A}$ | $1.0^{A}$ | $1.0^{A}$ |
| Lagodon rhomboides | Island | 25 m | $0.5^{AB}$ | $0.0^{C}$ | $0.0^{C}$ | $0.3^{BC}$ | $0.8^{A}$ | $0.8^{A}$ |
| Fundulus heteroclitus | Mainland | 100 m | 0.2 | 0.0 | 0.2 | 0.0 | 0.0 | 0.0 |
| Fundulus heteroclitus | Island | 100 m | 0.2 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Lagodon rhomboides | Mainland | 100 m | $0.7^{A}$ | $0.0^{B}$ | $0.0^{B}$ | $0.8^{A}$ | $1.0^{A}$ | $0.8^{A}$ |
| Lagodon rhomboides | Island | 100 m | $0.8^{A}$ | $0.0^{B}$ | $0.0^{B}$ | $0.6^{A}$ | $1.0^{A}$ | $1.0^{A}$ |
| Fundulus heteroclitus | Interior | Creek | 1.0 | 0.7 | 0.7 | 1.0 | 1.0 | 0.7 |
| Lagodon rhomboides | Interior | Creek | $0.5^{AB}$ | $0.0^{B}$ | $0.0^{B}$ | $0.5^{AB}$ | $0.5^{AB}$ | $0.8^{A}$ |

Fig. 2-1. Site locations within Bogue, Back and Core Sounds of North Carolina. Locations corresponding to the circles are mainland salt marsh sites, squares are large island salt marsh sites, stars small island salt marsh sites and triangles are interior salt marsh sites.

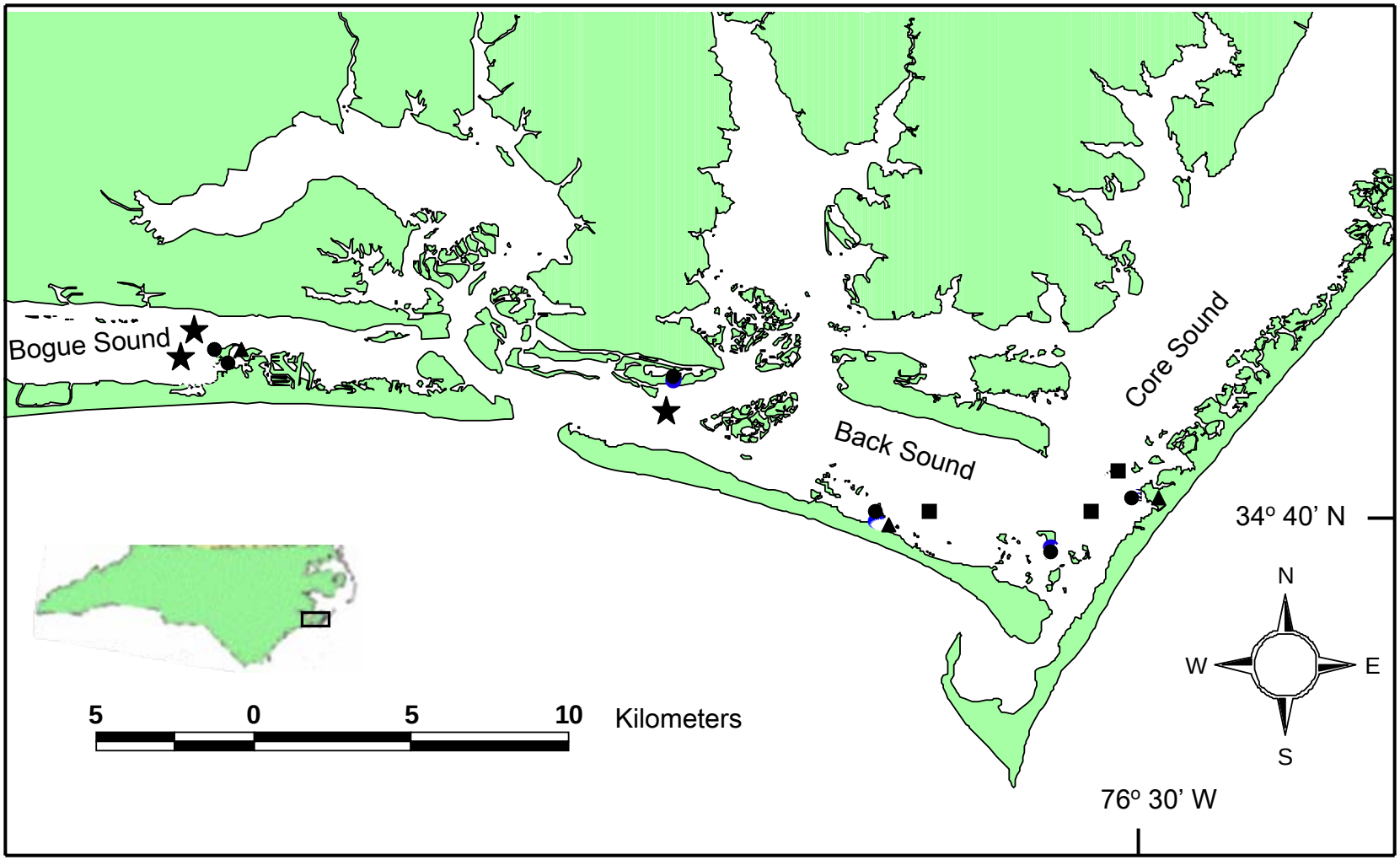
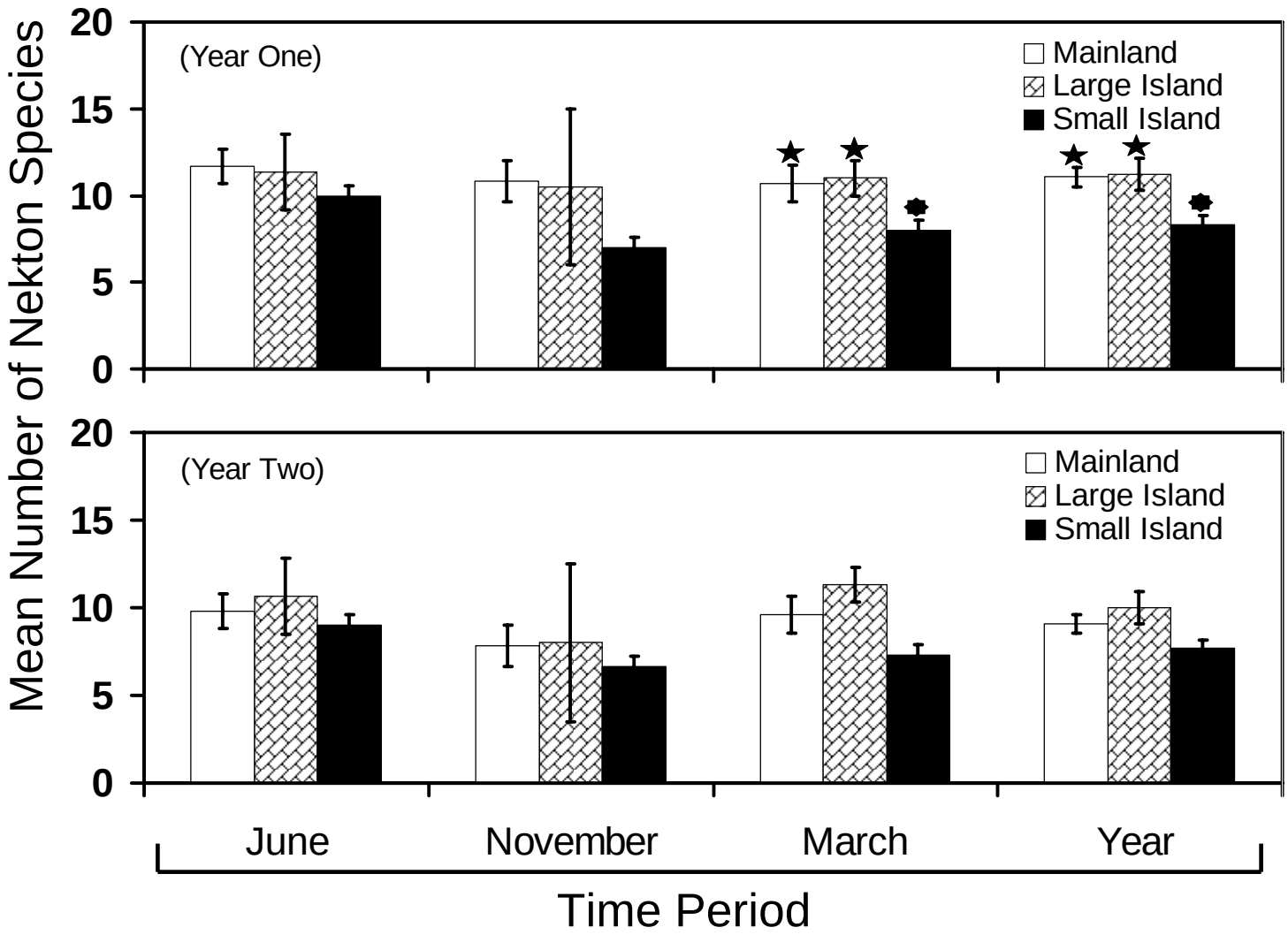
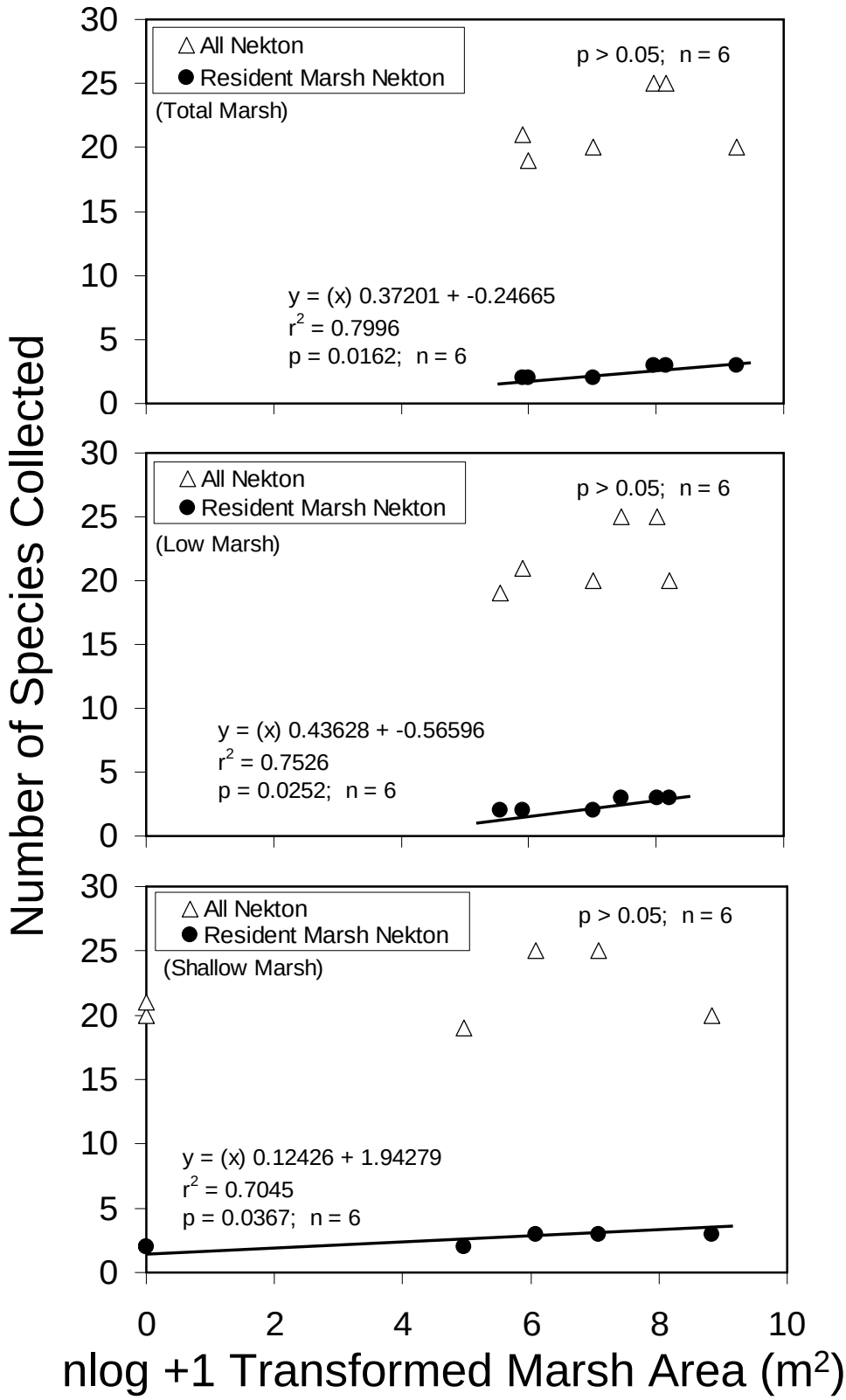

Bogue Sound
Back Sound
Core Sound
34° 40' N
76° 30' W
N
W
E
S
5
0
5
10
Kilometers

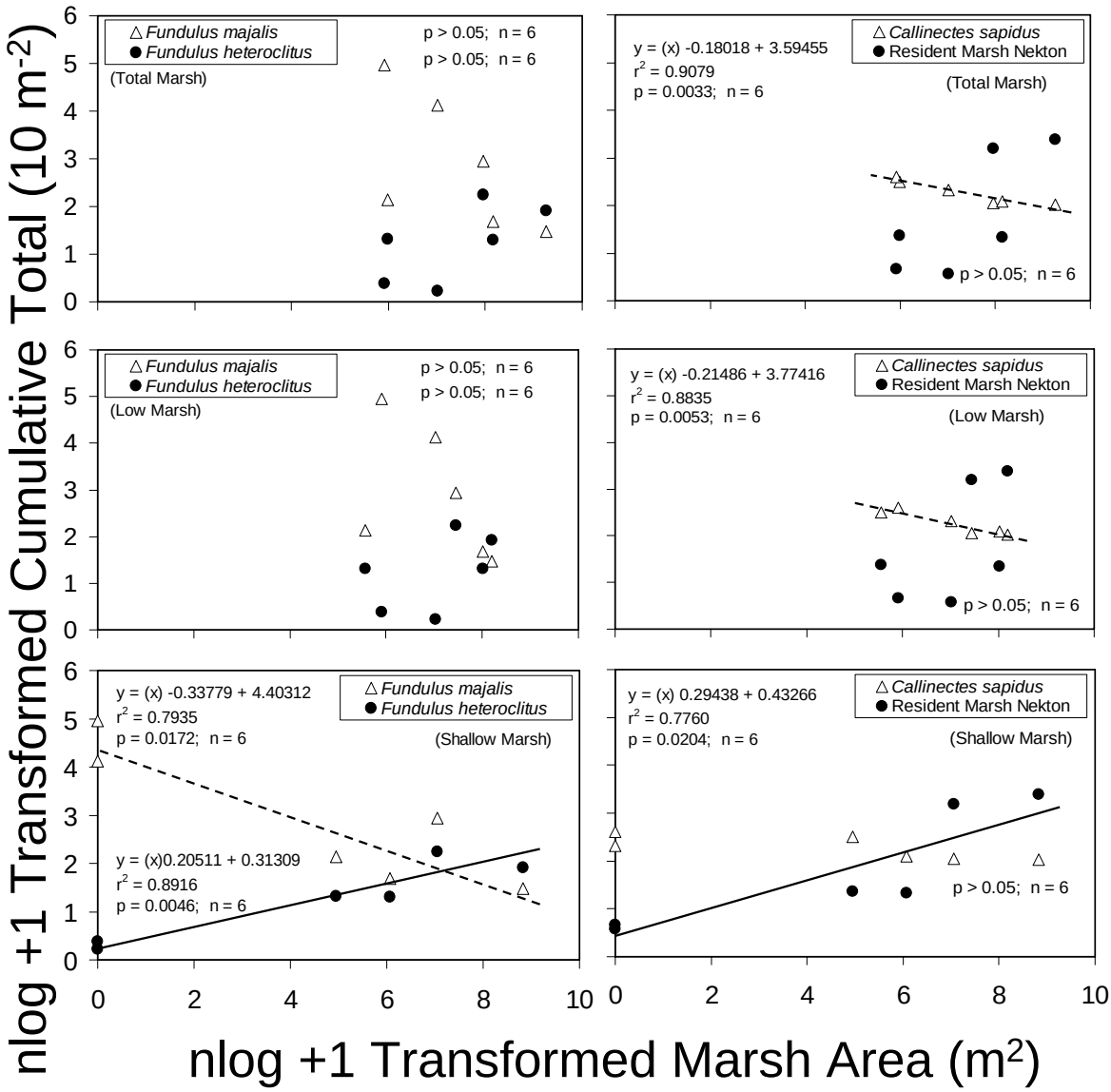
Fig. 2-2a,b. Mean number of nekton species per marsh type by sampling date and for each year as a whole. During each collection period, marsh types that significantly differ from one another ($p \leq 0.05$) are signified by a different symbol. Error bars correspond to one standard error.


Mean Number of Nekton Species
20
15
10
5
0
(Year One)
Mainland
Large Island
Small Island
20
15
10
5
0
(Year Two)
Mainland
Large Island
Small Island
June
November
March
Year
Time Period

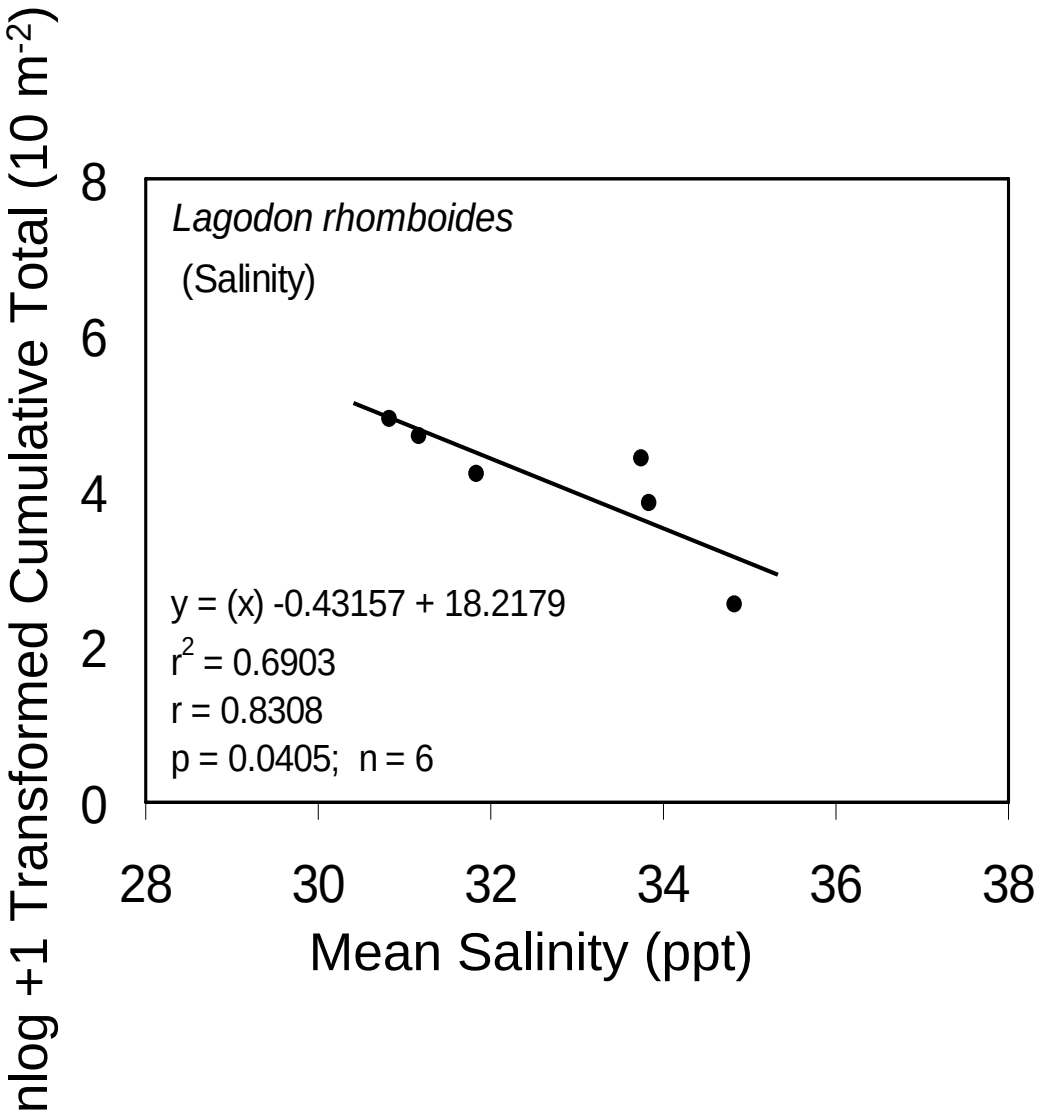
Figs. 2-3a-c. Linear regression analyses for cumulative number of resident marsh nekton and all nekton species combined per island site compared to nlog +1 transformed total, low and shallow marsh areal size, $m^2$, per island. N = 6 and $p \leq 0.05$ for slope significance level.


△ All Nekton
● Resident Marsh Nekton
(Total Marsh)
p > 0.05; n = 6
y = (x) 0.37201 + -0.24665
r2 = 0.7996
p = 0.0162; n = 6
△ All Nekton
● Resident Marsh Nekton
(Low Marsh)
p > 0.05; n = 6
y = (x) 0.43628 + -0.56596
r2 = 0.7526
p = 0.0252; n = 6
△ All Nekton
● Resident Marsh Nekton
(Shallow Marsh)
p > 0.05; n = 6
y = (x) 0.12426 + 1.94279
r2 = 0.7045
p = 0.0367; n = 6
Number of Species Collected
0
5
10
15
20
25
30
0
2
4
6
8
10
nlog +1 Transformed Marsh Area (m2)

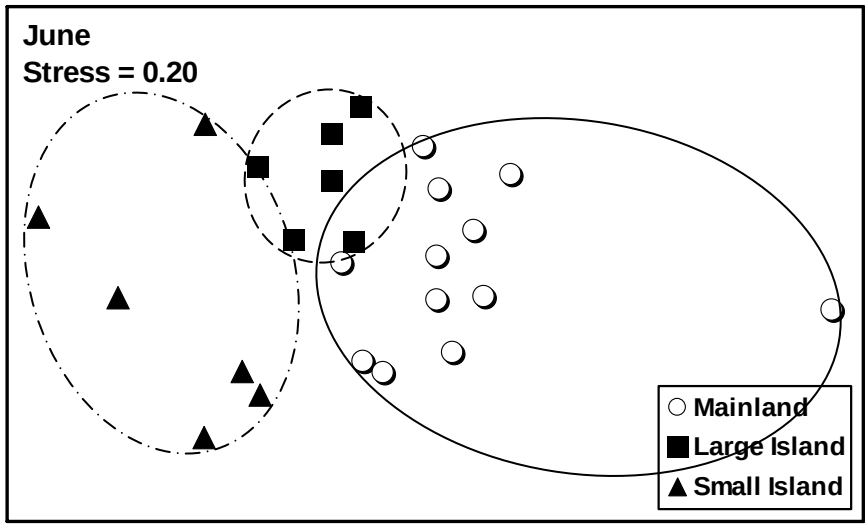
Figs. 2-4a-f. Linear regression analyses for nlog +1 transformed cumulative number of Fundulus heteroclitus, Fundulus majalis, Callinectes sapidus and all resident marsh nekton combined 10 $m^{-2}$ per island site compared to nlog +1 transformed total, low and shallow marsh areal size, $m^2$, per island. N = 6 and $p \leq 0.05$ for slope significance level.

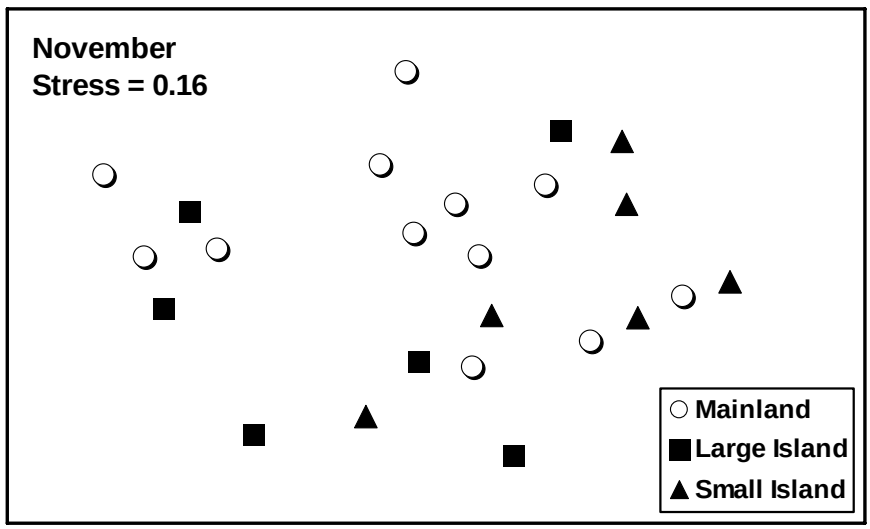

△ Fundulus majalis
● Fundulus heteroclitus
(Total Marsh)
p > 0.05; n = 6
p > 0.05; n = 6
△ Callinectes sapidus
● Resident Marsh Nekton
(Total Marsh)
y = (x) -0.18018 + 3.59455
r2 = 0.9079
p = 0.0033; n = 6
p > 0.05; n = 6
△ Fundulus majalis
● Fundulus heteroclitus
(Low Marsh)
p > 0.05; n = 6
p > 0.05; n = 6
△ Callinectes sapidus
● Resident Marsh Nekton
(Low Marsh)
y = (x) -0.21486 + 3.77416
r2 = 0.8835
p = 0.0053; n = 6
p > 0.05; n = 6
△ Fundulus majalis
● Fundulus heteroclitus
(Shallow Marsh)
y = (x) -0.33779 + 4.40312
r2 = 0.7935
p = 0.0172; n = 6
y = (x)0.20511 + 0.31309
r2 = 0.8916
p = 0.0046; n = 6
△ Callinectes sapidus
● Resident Marsh Nekton
(Shallow Marsh)
y = (x) 0.29438 + 0.43266
r2 = 0.7760
p = 0.0204; n = 6
p > 0.05; n = 6
nlog +1 Transformed Cumulative Total (10 m-2)
nlog +1 Transformed Marsh Area (m2)
0
1
2
3
4
5
6
0
2
4
6
8
10

Figs. 2-5. Linear regression analyses for nlog +1 transformed cumulative number of Lagodon rhomboides 10 $m^{-2}$ per island site compared to mean salinity (ppt) per island. N = 6 and $p \leq 0.05$ for slope significance level.


nlog +1 Transformed Cumulative Total (10 m-2)
8
6
4
2
0
Lagodon rhomboides
(Salinity)
y = (x) -0.43157 + 18.2179
r2 = 0.6903
r = 0.8308
p = 0.0405; n = 6
28
30
32
34
36
38
Mean Salinity (ppt)

Figs. 2-6a-c. Non-metric multidimensional scaling ordinations of square root transformed taxa abundance data collected from mainland, large island and small island salt marshes during June, November and March time periods for years one and two combined.


June
Stress = 0.20
Mainland
Large Island
Small Island


November
Stress = 0.16
Mainland
Large Island
Small Island

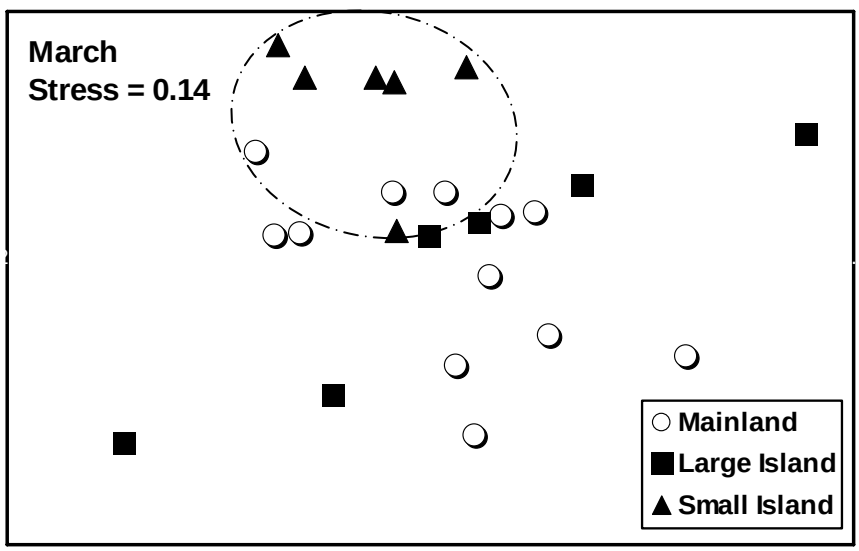
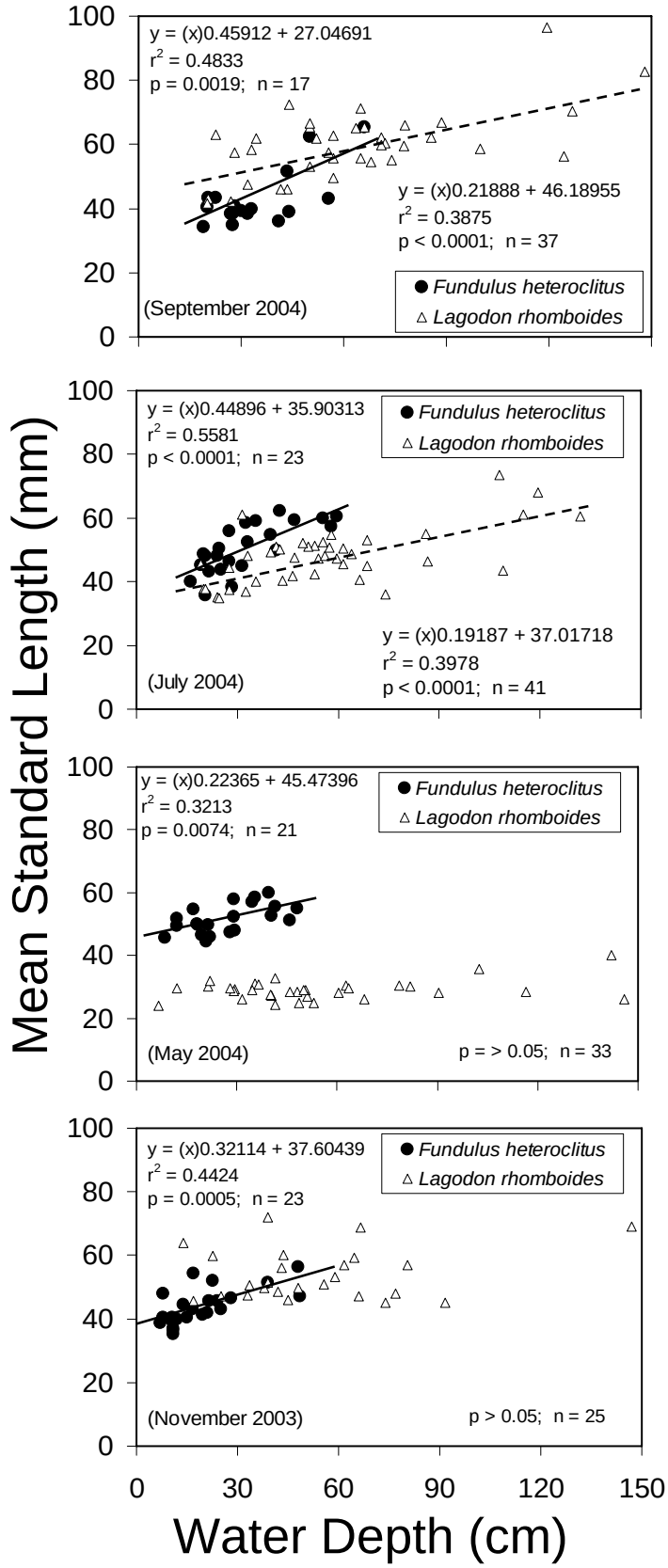

March
Stress = 0.14
Mainland
Large Island
Small Island

Figure 2-7. Linear regression analyses for non-isolated salt marshes (mainland and interior salt marshes combined) comparing fish mean standard length to average eel pot water depth for all placement locations during the four collection time periods (November 2003, May 2004, July 2004 and September 2004) when Fundulus heteroclitus and Lagodon rhomboides were most abundant.


Mean Standard Length (mm)
Water Depth (cm)
y = (x)0.45912 + 27.04691
r2 = 0.4833
p = 0.0019; n = 17
y = (x)0.21888 + 46.18955
r2 = 0.3875
p < 0.0001; n = 37
Fundulus heteroclitus
Lagodon rhomboides
(September 2004)
y = (x)0.44896 + 35.90313
r2 = 0.5581
p < 0.0001; n = 23
y = (x)0.19187 + 37.01718
r2 = 0.3978
p < 0.0001; n = 41
(July 2004)
y = (x)0.22365 + 45.47396
r2 = 0.3213
p = 0.0074; n = 21
(May 2004)
p = > 0.05; n = 33
y = (x)0.32114 + 37.60439
r2 = 0.4424
p = 0.0005; n = 23
(November 2003)
p > 0.05; n = 25
0
20
40
60
80
100
0
30
60
90
120
150

# CHAPTER 3.

# NEKTON RECRUITMENT AND PREDATION POTENTIAL WITHIN ISOLATED ISLAND SMOOTH CORDGRASS (SPARTINA ALTERNIFLORA) SALT MARSH VERSUS NON-ISOLATED SALT MARSH.

# INTRODUCTION

Previous work (Chapter 2) has shown that island salt marshes function differently from mainland salt marshes and do not support equivalent populations or densities of resident salt marsh nekton compared to those larger salt marsh areas. The size of island salt marshes and sub-habitats can also significantly affect nekton usage patterns and minimal size thresholds have been observed for support of resident salt marsh species (Chapter 2). Recent research also suggests that local populations of nekton with limited dispersal ability, such as the salt marsh resident mummichog (Fundulus heteroclitus), can be driven to extinction by collection without replacement, while populations of more transient species with good dispersal ability, such as pinfish (Lagodon rhombodies), might be little affected (Meyer et al. 1996). An underlying question is why are these disparities in utilization patterns present and what is responsible for their occurrence?

Habitat proximity to other, similar habitats regardless of size, and habitat size regardless of degree of isolation from like habitats, has been considered important for recruitment of new individuals and the number of species that can be supported (Rieman and McIntyre 1995; Acosta 1999; Lockwood et al. 2002; Chapter 2). Terrestrial research suggests that growth, production and resilience of populations with limited dispersal ability in small isolated patches may be lower than that of similar-sized habitat with connectivity to expansive habitat, or to the larger habitat unit itself (Fahrig and Merriam 1985). The size of discrete habitat patches can significantly influence the intrinsic rate of population growth (recruitment) for species with low vagility (MacArthur and Wilson, 1967; Harrison and Taylor 1997) and determine source/sink aspects for individual species (Hanski 1994). For species such as Fundulus heteroclitus, habitat size and the

occurrence of sufficient shallow habitat to elude predators (Posey and Hines 1991; Ruiz et al. 1993; Chapter 2) can be important for population maintenance. The importance of refuge habitats might be particularly important in areas that are situated in open water and frequented by predator species (Rountree and Able 1997) including salt marsh islands.

Often numerically dominant within salt marshes, Fundulus heteroclitus is considered to have restricted dispersal ability with an observed home range of less than 400 m (Lotrich 1975; Abrams 1985). This species is entirely dependant on salt marsh (Chapter 2) in which they spawn and lay eggs upon Spartina alterniflora stems (Taylor and DiMichele 1983). Lagodon rhomboides, by contrast, is a pelagic spawner that recruits within estuaries and utilizes multiple habitats (Fonseca et al. 1990; 1996; Meyer et al. 1996; 1999; Hovel et al. 2002; Paperno et al. 2001) including salt marshes (Hettler 1989; Meyer et al. 1996). Differences in these life history aspects suggest that these two species might not only differ in recruitment potential, but also in susceptibility to local population extinctions (Chapter 2), impacted to a significant degree by predation vulnerability within the expanse of habitats between salt marshes (Heck and Thoman 1981; Ruiz et al 1993), particularly within island salt marshes.

The population size of species with limited dispersal ability that reside within isolated island salt marshes may be more adversely affected by lack of resources, including sufficient critical refuge habitat, than non-isolated habitats, and hence habitat carrying capacity might be lower. Effects may be particularly evident for created salt marsh habitats that are initially simple in terms of habitat complexity and faunal diversity (Minello and Zimmerman 1992; Sacco et al. 1994; Levin et al. 1996). Abundances of

small resident species in small isolated salt marshes might be lower than non-isolated salt marshes, and these populations might have less resiliency to ecological perturbations (Chapter 2).

Salt marsh restoration projects within the Atlantic coast of the USA have typically involved the creation of low marsh comprised of smooth cordgrass (Spartina alterniflora) < 4,000 $m^2$ in size (Matthews and Minello 1994). Increasingly, restorations have targeted isolated island salt marsh habitats (Meyer 2001) to reduce anthropogenic habitat impacts and define legal ownership issues. However, design of such isolated island restoration projects needs to consider whether the habitat will act as a source or a sink for target species populations. Does the size and the location of the restoration influence population maintenance for species with limited dispersal ability? These are key factors when considering the creation and restoration of habitat. One might expect that estuarine salt marsh habitat would act similar to terrestrial habitats for dispersal-restricted species, and that small habitats with very little connectivity to like habitats would be prone to local extinctions and exhibit a mainland-island pattern of utilization with the mainland habitats providing colonizers for smaller islands whose populations face periodic extinction (Harrison and Taylor 1997). However, little information is available for estuarine nekton populations to determine applicability of terrestrial-based population theories.

The overall objective of this current research was to determine whether size and degree of isolation of salt marsh habitat affects its function for nekton support. This study examined recruitment, potential predation impacts, and population dynamics for two nekton species that represent differing dispersal abilities, Fundulus heteroclitus (limited

ability) and Lagodon rhombodies (good ability), both co-dominants within intertidal salt marshes along the South Atlantic coast of the USA (Hettler 1989; Meyer et al. 1996; Chapter 2). A combination of approaches were used including measurement of annual recruitment potential, predation-related mortality potential and habitat quality influence at replicate salt marsh sites.

## MATERIALS AND METHODS

### Sites

The six naturally occurring island salt marshes that were paired with six mainland salt marshes and the interior salt marshes for the various collections (see Chapter 1) were located amongst shallow water flats composed of a seagrass and unvegetated bottom mosaic within Bogue, Back and Core Sounds of North Carolina, USA (Fig. 3-1). Island salt marsh sites were typically at least 400 m away from the nearest other salt marsh.

### Environmental Parameters

Salinity, water temperature, marsh fringe elevation and marsh slope were measured for each mainland and island salt marsh site during concurrent fyke and block net (see Chapter 2) and gill net collections for each collection period. Both salinity (measured with a temperature compensated refractometer, accuracy of 0.1 ppt) and temperature were measured once nets were set.

Salt marsh vertical range was measured at each site, as were area size, perimeters of the salt marsh sites and distance from island salt marshes to the nearest other salt marsh (see Chapter 2). Shallow and low salt marsh sub-habitats were also delineated

(see Chapter 2). Mainland perimeter and low salt marsh area estimates used to compare predator access potential were delineated using shallow salt marsh and other ecological boundaries including adjacent creek and geographic points.

Marsh Vegetation

At each site, vegetation attributes were measured by stem density counts and morphometrics, based on methodology used by Cowie et al. (1992) and Meyer et al. (2001). Measurements were performed concurrently with fyke and block net collections (see Chapter 2).

Larval and Juvenile Fish Collection

From May-August 2003, the peak annual recruitment period for resident marsh species including Fundulus heteroclitus (Brummett 1966; Talbot and Able 1984; Abrams 1985), larval (individuals less than 16 mm total length) and juvenile (individuals greater that or equal to 16 mm and less than 37 mm total length) (Abrams 1985) fish collections were conducted targeting F. heteroclitus using modified Breder traps (Breder 1960) and pit traps (Able and Hagen 2000). These collections were conducted monthly at each of the 12 fringing salt marsh sites selected for fyke net collections (isolated small and large island salt marshes and mainland salt marsh sites), and at three interior salt marsh sites. During these monthly collections, paired modified Breder traps, constructed out of clear 'Plexiglass', 45 cm length x 20 cm height x 30 cm width, with duel front and back V-shaped collection funnels tapering to 8 mm collections slots, were positioned within the sites delineated for fyke net collections (Chapter 2). For interior salt marshes, areas

sampled were similar to fyke net collection areas at the other sites. During low flood tides, one Breder trap was anchored to the sediment in the low salt marsh ~ 3 m from the salt marsh edge, and one midway between the salt marsh fringe and the back of the site on opposite sides of the 10 m wide delineated area. Breder trap openings were oriented towards the salt marsh fringe into the perceived flood tide current direction so that fish moving into the salt marsh during the flood tide were collected. Breder traps were allowed to fish until the trap top was covered by water and then retrieved (~2 h duration).

Pit traps were also used for collection of larval and juvenile Fundulus heteroclitus (Able and Hagen 2000), consisting of plastic funnels 14.5 cm in diameter by 12.5 cm deep at the tapered point. Paired pit traps were set flush to the sediment surface on opposite sides of the 10 m wide delineated sampling site at three distances, two meters inside of the salt marsh fringe, midway between the salt marsh fringe and the back of the site, and within the high salt marsh, at each of the 15 sites (including interior salt marsh sites). Pit traps were held in place by 170 g lead fishing weights, which also were used to plug the bottom holes of the funnels. All pit traps were set at low tide one tidal cycle prior to collection, and contents collected during a subsequent low tide.

The contents for each Breder and pit trap were emptied onto a 0.5 mm mesh sieve and all fauna collected preserved in 95 % ethanol. All fishes, shrimps and crabs collected were identified to species, enumerated and measured for each Breder and pit trap (SL and total length for fishes, total length for shrimps and carapace width for crabs).

Due to low catch numbers per individual Breder and pit trap, sparse monthly catches were observed for most sites. Because of these sparse catches and the fact that this sampling entailed equal sampling effort at each site for the different salt marsh types

examined, the total number of individuals for both larval and juvenile categories of F. heteroclitus were pooled on a per site bases for the summer for all Breder and pit traps combined (n = 3 for interior, large and small island salt marshes and n = 6 for mainland salt marsh types).

Marsh Nekton Collection

During years one (June 2002-March 2003) and two (June 2003-March 2004), nekton were collected within each salt marsh treatment pair to examine utilization patterns among the different salt marsh types (especially for the two target species, Fundulus heteroclitus and Lagodon rhomboides) during different life history stages. Collections occurred during late spring/early summer (June), a time of peak young of year (YOY) recruitment for F. heteroclitus and when L. rhomboides have attained juvenile size class status; the mid fall (November), a time when YOY F. heteroclitus have attained juvenile or adult size and L. rhomboides have attained adult size; and late winter/early spring (March), a time when L. rhomboides YOY recruit to estuarine habitats and adult F. heteroclitus prepare to spawn within the salt marshes. Fyke and paired block nets constructed of black 3.2 mm mesh were used for marsh faunal collection (McIvor and Odum 1986; 1988; Meyer et al. 1996; 2001). Fyke nets with a 1 $m^2$ mouth and 5.5 m wings, in combination with 30 m block nets, were used at each site to collect marsh nekton (see Chapter 2).

Nekton collected were identified to species, measured on site and released live. For each site the number of individuals and wet-weight biomass for each species were recorded. All of the individuals for a species were measured or, if numerically abundant,

a randomly selected subsample of at least 100 individuals, or 5 % of the total (whichever was higher), were measured (standard length (SL) for fishes, total length for shrimps and width, tip to tip of anterolateral carapace spines, for crabs).

Marsh Edge Nekton Collection

Low salt marsh fringe use by large fishes, not typically collected using fyke nets, was examined during year one (June 2002-March 2003) using nocturnal gill net sampling (Rountree and Able 1997), for large, small and mainland salt marsh types during summer, fall and winter/spring sampling dates. Initially, this collection involved the use of 10 m length x 1.8 m tall paired gill nets, one constructed of 6.4 and the other with 14.0 cm stretch monofilament mesh. Paired gill nets were set up at the edge of the salt marsh fringe in an L-shaped configuration with the net extending 5 m seaward, perpendicular to the fringe, and 5 m parallel to the marsh fringe with the open portion of the L facing upstream during flood tide. A 5 m buffer separated the different sized gill net panels with the larger mesh positioned up current of the smaller mesh on flood tide. Nocturnal collections were performed because of decreased gear avoidance by fish versus daylight collections (Rountree and Able 1997). Nets were erected with attachment poles during evening low tides, allowed to fish over one complete tidal cycle, and retrieved during the subsequent morning low tide. During the second year (June 2003-March 2004), gill net collection was modified to include nekton collection within the salt marsh and involved the use 10 m length x 1.8 m tall paired gill nets, one constructed of 6.4 and the other with 14.0 cm stretch monofilament mesh set in a ⌐⌐ type configuration. As during

year one, the lower portion of the gill nets were facing upstream during flood tide, again with the larger mesh upstream of the smaller mesh net. However, during year two an additional, 10 m length x 1.8 m high, marsh gill net section (inside salt marsh) for both sized nets were placed in a pattern opposite that of the lower section of gill net (outside marsh fringe). These additional gill nets were set to extend from the salt marsh edge 4 m landward, perpendicular to the fringe, and 6 m parallel to the salt marsh fringe with the open portion of the gill net facing upstream during ebb tide. These additional net sections were added to examine potential differences in predatory fish incursion into the salt marshes. During year two, gill nets were similarly erected with attachment poles during evening low tides, allowed to fish over one complete tidal cycle, and retrieved during the subsequent morning low tide. All fish collected during both year one and two collections were identified and measured, (SL) and wet weight for each fish recorded.

Statistical Analysis

Because of the preponderance of zero catches, and low replication (n = 3 for large and small island, and interior salt marshes), comparisons between the different salt marsh types utilized a conservative distribution-free single classification method of analysis, the Kruskal-Wallis test (Sokal and Rohlf 1981), to test for distribution differences in larval and juvenile Fundulus heteroclitus from Breder and pit trap, and fyke and block net collections, and for Lagodon rhomboides based on fyke and block net collections. The Kruskal-Wallis test was also used to compare vegetation stem density estimates for each salt marsh type during each sampling date as well as physical measures for each site during each year. Replication level was the site. Gill net catch

data were analyzed based on individuals collected per set. Kruskal-Wallis tests were also used to test for nekton gill net abundance differences between salt marsh types for each sampling period, for the year as a whole, between intra-annual sampling periods and intra-marsh area collections during year two. Predation potential comparisons for the different salt marsh types was assessed using analysis of variance (ANOVA) and post hoc Student Neuman-Keuls multiple range test (Sokal and Rohlf 1981), relating low salt marsh area and low salt marsh access perimeter estimates (area/perimeter) and also cumulative predator catch per site via gill net collections compared to area/perimeter estimates [(area /perimeter)/cumulative predator catch]. Data used for predation potential estimates were tested for homogeneity of variances using the $F_{max}$ test (Sokal and Rohlf 1981). If variances were found not to be homogenous, data were ln (x + 1) transformed to improve homogeneity of variance and again tested to assure data conformity.

Linear Regression Analysis (regression analysis) was utilized to examine YOY and adult cohort densities for Fundulus heteroclitus and Lagodon rhomboides from fyke net collections, and nekton gill net cumulative catch per unit effort (CPUE) in relation total salt marsh size and area of shallow and low salt marsh sub-habitats. Additionally, regression analysis was utilized to examine the relationship of potential predation pressure of the different sites for observed and equivalent (equal CPUE values substituted for all sites) cumulative predator gill net CPUE based on low marsh area to low marsh perimeter ratios, and observed cumulative predator CPUE versus YOY and adult cohort abundances of F. heteroclitus and L. rhomboides. Data used in regression analysis were tested for normality using the Shapiro-Wilk test (Shapiro and Wilk 1965; Sen et al. 2003). If data were not normal, data were ln (x + 1) transformed and again

tested to assure data conformity. For all data analyses minimal significance level was $p = 0.05$.

## RESULTS

### Environmental Parameters

As selected for, consistent physical parameter patterns that were present among the marsh types and did not change from year one to year two included habitat dimension characteristics and location (Table 3-1). The amount of total, low and shallow salt marsh areas were significantly different among mainland, large island and small island salt marsh types, and roughly an order of magnitude difference was observed between salt marsh types (Table 3-1). While large and small island salt marsh types were significantly farther away from like habitat than were mainland salt marshes, there was no significant difference in distance among different island types (Table 3-1). The vertical range for the different salt marsh types indicated significantly wider ranges of vegetation occurred in island types than for mainland salt marshes (Table 3-1). Consistently, though not significant, small island salt marshes had lower salt marsh fringe elevation and steeper marsh slope during both year one and two, while mainland salt marshes had a shallower fringe and marsh slope. Salinity was relatively consistent among years and salt marsh type. Water temperature at the island salt marsh sites was more variable than at the mainland sites among years, and even among salt marsh island type, with significantly higher mean annual temperature observed at the small compared to the large island salt marshes during year two, with the opposite pattern during year one (Table 3-1).

Vegetation Parameters

Smooth cordgrass (Spartina alterniflora) dominated the salt marsh vegetation in all areas sampled for this study. Other species were relatively rare, though glass wort (Salicornia virginica) was occasionally observed within mainland and large island salt marshes (Table 3-2). Total vegetation and S. alterniflora stem density was consistently lower at large island salt marshes compared to both mainland and small island salt marshes for the year as a whole and during most collection periods during both years, significantly during March 2003 (Table 3-2). A consistent though typically non-significant pattern of higher densities of standing dead S. alterniflora stems was observed within the mainland compared to both large and small island marshes, with significantly higher densities during June 2003 within the mainland and large island salt marshes compared to small island salt marshes. However, no clear pattern was present relative to dead S. alterniflora stem density between large and small island salt marshes (Table 3-2).

Larval and Juvenile Fish

Cumulative Fundulus heteroclitus larval abundance from Breder and pit trap collections differed significantly between salt marsh types. Interior salt marsh contained significantly more larvae than did all other salt marsh types (Fig. 3-2). Fundulus heteroclitus larval abundance for mainland and large island salt marsh types were similar to one another ($p > 0.05$) and both were significantly greater than that for small island salt marsh (Fig. 3-2). Juvenile F. heteroclitus abundance was highest within the interior followed by mainland salt marsh, both of which differed significantly from large and small island salt marsh types, which had few juvenile F. heteroclitus during the summer

collections (Fig. 3-2).

Nekton Size Versus Salt Marsh Type

A distinct YOY Fundulus heteroclitus cohort size class was apparent from block and fyke net collections, including individuals < 32 mm SL (37 mm total length) during June and March and < 51 mm SL during November. Fundulus heteroclitus YOY were consistently more abundant within mainland compared to large and small island salt marshes, and within large island compared to small island salt marshes, during both years. Significant differences in YOY abundance were observed during June of both years (mainland > large and small island) and during November of year two (mainland > small island). Significant differences were also observed for YOY cumulative totals for year one (mainland > large and small island) and year two (mainland > small island) (Figs. 3-3a,b). Year one and older (Y1, Y1+) F. heteroclitus abundance patterns were similar to those observed for YOY with Y1, Y1+ abundance consistently higher within mainland compared to large and small island salt marshes, and higher abundances within large island compared to small island salt marshes, during both years. Significant differences in F. heteroclitus Y1, Y1+ abundance were observed during June year one (mainland > small island) and for cumulative totals for both year one (mainland and large island > small island) and year two (mainland > large and small island) (Figs. 3-3a,b).

Young of year abundances for Lagodon rhomboides were similar among salt marsh types and were consistently dominated by YOY (those individuals < 30 mm SL during March, < 50 mm SL during June, and < 80 mm SL during November) during all collection periods at each salt marsh type. No consistent differences in YOY or Y1, Y1+

abundance was evident among the salt marsh types during any collection period, or for cumulative abundance for either year (Figs. 3-4a,b).

Linear regression analysis revealed that cumulative YOY Fundulus heteroclitus abundance significantly increased with increase in total salt marsh area and shallow salt marsh area (Figs. 3-5a-c). However, no significant relationship was observed with island low marsh area (Fig. 3-5b). No significant linear regressions occurred for cumulative abundances of Y1, Y1+ F. heteroclitus, or YOY and Y1, Y1+ Lagodon rhomboides with island salt marsh habitat area estimates. No significant relationship was apparent between cumulative abundance of F. heteroclitus and L. rhomboides size class cohorts and gill net cumulative CPUE of nekton feeding fishes (nektonivores).

Marsh Edge Nekton Collection

A total of 24 different nekton species were collected during gill net sampling. No significant differences were evident between marsh type and the number of nekton species collected during either year (Table 3-3). During both years the highest number of nekton species encountered consistently occurred during June for all salt marsh types.

Cumulative annual gill net CPUE was generally higher within small and large island salt marsh compared to mainland salt marsh during both years for nektonivorous and non-nektonivorous fishes (Figs. 3-6a,b), though no significant differences were apparent. Examination of data by collection date similarly indicated a pattern of higher CPUE for island salt marsh types compared to mainland salt marsh with significant differences evident for large island salt marsh compared to both small island and mainland salt marsh during March of year two for all fish feeding mode combinations

(Figs. 3-7a,b).

When gill net CPUE was considered by collection date, regardless of salt marsh type, year one CPUE was consistently higher among all feeding modes during June compared to November and March, with significantly higher CPUE during June compared to November for all feeding modes combined and CPUE for non-nektonivorous fishes significantly higher during June than both November and March (Fig. 3-8a). Year two CPUE for June and March were higher than that observed for November for all feeding modes, and nektonivorous fish CPUE was significantly higher during June than both November and March. June CPUE for non-nektonivorous and all feeding modes was significantly higher than during November (Fig. 3-8b).

Based on collection date and salt marsh type during year one, CPUE was consistently higher among all feeding modes during June compared to November and March for all salt marsh types. Significant differences were evident between June compared to November and March for non-nektonivorous fishes at large island salt marshes (Fig. 3-9a). During year two, significantly higher CPUE was observed for large island salt marsh during both June and March compared to November for all feeding categories (Fig. 3-9b), and for small island salt marsh during June compared to both November and March for all fish feeding modes combined and for nektonivorous fishes (Fig. 3-9b).

Gill net CPUE inside salt marsh habitat (the 5 m x 5 m L shaped section of gill net located inside the salt marsh) versus outside salt marsh habitat (the 5 m x 5 m L shaped section of gill net located along the outside edge of the salt marsh) revealed no significant differences (Fig. 3-10). Comparisons between salt marsh types for cumulative

annual outside and inside CPUE also indicated no significant differences (Figs. 3-11a,b).

During both years there was higher CPUE for species known to be pelagic or open-water oriented, including Atlantic menhaden (Brevoortia tyrannus), ladyfish (Elops saurus), smooth dogfish (Mustelus canis) and butterfish (Peprilus triacanthus), in island salt marshes than mainland salt marshes (Table 3-4). While many nekton species were collected by gill nets, catches among salt marsh types were not consistent, making it difficult to distinguish patterns. Those that were evident included significantly higher CPUE for E. saurus during June 2002, and the year as a whole, within small island compared to both large island and mainland salt marsh; significantly higher cumulative CPUE for the year as a whole for spot (Leiostomus xanthurus) within small island salt marsh compared to mainland salt marsh; and significantly higher cumulative CPUE for the year as a whole for gulf flounder (Paralichthys albigutta) within large island salt marsh compared to mainland salt marsh (Table 3-4). During year two CPUE differences were only apparent for L. xanthurus for which significantly higher CPUE was apparent within the large island salt marsh compared to mainland and small island salt marsh during March (Table 3-4).

During both years, most nekton species were more abundant during June than during November or March (Table 3-5). When CPUE for individual species was compared among collection dates for salt marsh types, significant differences were apparent with higher CPUE during year one for Lagodon rhomboides within mainland salt marsh during June compared to both November and March, and for Mugil cephalus within large island salt marsh during June compared to both November and March (Table 3-5). Similarly, during year two significantly higher CPUE was observed for southern

flounder (Paralichthys lethostigma) within mainland salt marsh during June compared to both November and March (Table 3-5).

Low marsh accessability (area/perimeter) estimates indicated that small and large island salt marshes were four and two times, respectively, more accessible to large nekton, including nektonivorous species, than mainland salt marsh, with significant differences in potential accessability apparent between small island and mainland salt marsh (Table 3-6). When observed nektonivore abundances were considered to estimate potential predation influence, small and large island salt marshes had an order of magnitude higher predator : area/perimeter relationship than mainland salt marsh, and significant differences were apparent between both small and large island salt marsh compared to mainland salt marsh (Table 3-6). Linear regression analysis revealed a similar increase in predation potential simply based on the observed decrease in low salt marsh area/perimeter ratio among the sites, with a significant regression observed when cumulative nektonivore CPUE was maintained at equivalent levels at all salt marsh sites (the mean overall CPUE) (Figs. 3-12a,b). However, no significant linear relationship was apparent when actual cumulative nektonivore CPUE to salt marsh area/perimeter was represented in the analysis (Figs. 3-12a,b).

## DISCUSSION

It was evident that salt marsh usage patterns for the resident, Fundulus heteroclitus, and the transient, Lagodon rhomboides, differed significantly. While F. heteroclitus abundance for both YOY and Y1, Y1+ cohorts tended to show consistent differences among mainland, large island and small island salt marsh types, the

abundance of L. rhomboides was similar among the different salt marsh types for these cohorts. Differences in life history strategies exemplified by these two estuarine species may explain different distribution patterns. Lagodon rhomboides is a transient nekton species, whose pelagic-spawned larvae enter inlets (Warlen and Burke 1990) and disperse among the various estuarine habitats (Fonseca et al. 1990; Wenner et al. 1996; Paperno et al. 2001), using salt marshes only facultatively (Hettler 1989; Meyer et al. 1996). Fundulus heteroclitus is a species that only spawns within salt marsh habitat (Taylor et al. 1977), and is dependant upon salt marsh habitat throughout its entire life (Abrams 1985). It is relatively limited in dispersal ability (Meredith and Lotrich 1979) and in its use of adjacent shallow water habitats (Rozas and Odum 1987; Kneib and Wagner 1994).

Differences in Fundulus heteroclitus densities for the different salt marsh types are likely due to F. heteroclitus larval and adult dispersal capabilities, and reduced connectivity to source habitats, as was also shown by Armsworth (2002) for reef fish metapopulations. In contrast, Lagodon rhomboides did not exhibit distributions indicative of isolation effects. These conclusions are further supported by significantly lower densities of adult and juvenile F. heteroclitus within the large and small island salt marshes compared to mainland salt marsh, with no such patterns apparent for L. rhomboides. However, Breder and pit trap collections of larval and juvenile F. heteroclitus suggested that larval supply at the isolated large island and mainland salt marsh habitats was similar, and that both of these habitat types provided spawning sites for resident populations. Hence larval supply was not the primary factor limiting F. heteroclitus densities at the isolated large island salt marshes. It did, however, appear to

be the case at isolated small island salt marshes, possibly due to lack of breeding populations within the isolated small island habitats.

Disparities in larval Fundulus heteroclitus supply evident between interior salt marshes and all fringing salt marsh types (mainland, large and small island), and between small island and both mainland and large island salt marshes, may be explained by adult breeding population abundances. Breeding populations were more abundant within the interior and mainland salt marshes and least abundant within the small island salt marshes (Chapter 2). Similar patterns for site specific F. heteroclitus breeding population abundance relative to YOY production has been noted by Kneib (1997). The consistency of the abundance patterns observed suggested that breeding stock abundance was partially responsible for the F. heteroclitus population size structure. The interior salt marshes sampled in this study more closely resembled the protected upper reaches of interior creek type salt marshes examined by Kneib and Wagner (1994) and Tupper and Able (2000) in terms of physical setting. These habitat types might provide optimal F. heteroclitus breeding habitat with relatively low predation threat due to low predator concentrations compared to shallow water areas close to open water habitat (Tupper and Able 2000).

Disparities in larval supply did not translate into differences for juvenile Fundulus heteroclitus abundance between interior and mainland salt marsh nor did similarities in larval supply translate into similarities in juvenile abundance among mainland and large island salt marshes. While juvenile movement from the interior to mainland fringing salt marshes could account for equivalent juvenile abundances, and juvenile emigration from large island salt marshes could cause reduced abundances compared to mainland salt

marshes, such explanations do not seem likely to account for the disparities in larval and juvenile F. heteroclitus patterns. Movement of larval and juvenile F. heteroclitus is restricted to salt marsh refuge (Kneib 1987; Kneib and Wagner 1994; Able and Hagan 2000; Raichel et al. 2003). A high marsh berm barrier at my sites separated interior from mainland salt marsh types and eliminated overwash during the astronomically low summer tides (Hutchinson and Sklar 1993), preventing mixing of salt marsh populations. Other factors, such as density-dependant cannibalism (Kneib 1987; Halpin 2000), might partially explain the disparity between larval and juvenile F. heteroclitus abundances observed for interior and mainland salt marsh. The differential pattern between larval and juvenile contribution to F. heteroclitus population for mainland and large island salt marshes suggests differential survival for YOY also occurred between these two salt marsh types.`

Similarities among mainland and isolated large island salt marsh habitats for larvae and subsequent disparity in juvenile Fundulus heteroclitus abundances suggest that a bottleneck constrained adult and juvenile F. heteroclitus populations at large island salt marshes. Further, these patterns suggest that population limitation for F. heteroclitus at the large island salt marsh habitats occurs with juveniles. Similar bottlenecks between the larval, juvenile and adult size classes for F. heteroclitus and have been postulated or shown to be due to absence of low tide refuge (Able and Hagan 2000; Able et al. 2003), reduced prey concentrations among habitat types (Weisberg and Lotrich 1986) or for specific size classes (Raichel et al. 2003), and competition for resources with sympatric species including Fundulus majalis (Baker-Dittus 1978; Weisburg 1986) which were more abundant within island salt marshes than F. heteroclitus (Chapter 2). However,

differential predation rates among the different salt marsh types might also have significantly contributed to disparities in YOY abundance.

Predator concentrations were consistently higher during peak spawning season for F. heteroclitus and during times of peak adult and juvenile F. heteroclitus occurrence. This temporal overlap of F. heteroclitus occurrence and occurrence of nekton predators that opportunistically feed upon F. heteroclitus, including blue fish (Pomatomus saltatrix) (Bigelow and Schroeder 1953; Buckel and Stoner 2004), weakfish (Cynoscion regalis) (Bigelow and Schroeder 1953), red drum (Sciaenops ocellata) (Abrams 1985), summer flounder (Paralichthys dentatus) (Meredith and Lotrich 1979; Abrams 1985; Roundtree and Able 1992), and Atlantic croaker (Micropogonias undulatus) (Ruiz et al. 1993), suggests that the highest predation pressure occurred during spawning and recruitment time periods for F. heteroclitus. Predation increase would be greatest within the island salt marsh habitats, due partly to greater predator accessibility and higher predator concentrations prevalent at these sites compared to mainland salt marsh. Predation would also disrupt migration between salt marsh patches, effectively isolating marsh-dependant species such as F. heteroclitus within separate salt marshes. Island salt marshes probably become most isolated from other salt marshes during the crucial periods of the breeding season, contributing to decreased population levels. This is similar to patterns observed for white-footed mice (Peromyscus leucopus) with regard to the size of isolated wood lots and population extinction susceptibility (Fahrig and Merriam 1985).

In addition to seasonal differences in predator abundance within the estuary, predator abundance and access (area/perimeter relationships) differed among salt marsh

types. The significant differences for these area/perimeter ratios among the salt marsh types suggests that island salt marshes may provide greater predator accessability compared to mainland salt marsh habitat. This would increase predation susceptibility for island populations of salt marsh dependant nekton such as F. heteroclitus, and significantly increase mortality compared to mainland salt marsh. This predation potential difference is analogous to the predation rate increase in bay scallops (Argopecten irradians) within fragmented seagrass habitats compared to contiguous seagrass habitats observed by (Irlandi 1994), and is representative of the general importance of predation refuge habitat for population maintenance (Harrison and Taylor 1997). Predation risk would be extreme for isolated resident populations located on island habitats where water levels top island salt marsh area, as typically occurs during daily tidal cycles at small island salt marsh and seasonally during periods of astronomically higher high spring tides (Hutchinson and Sklar 1993) at the large island salt marshes. Predation effects might be particularly intense within the small island salt marshes due to overall lack of high salt marsh habitat refuge for YOY, which could otherwise serve as predation refuge, as increased complexity in Spartina alterniflora basal plant morphology has been suspected to interfere with predation success on benthic prey in salt marsh habitats (Raichel et al. 2003).

The distribution of Fundulus heteroclitus at fringing mainland and island salt marshes was dependant on water depth with the highest abundance of both YOY and older individuals in the shallow salt marsh areas relative to deeper water salt marsh fringes (Chapter 2). Similar F. heteroclitus utilization patterns have been observed for shallow water habitats by Halpin (2000). Other investigators have suspected the

importance of shallow marsh habitat for F. heteroclitus populations (Able 1990) and YOY distribution (Talbot and Able 1984). The importance of shallow salt marsh habitat as a predation refuge for juvenile Fundulus heteroclitus is supported by linear regression analysis indicating a proportional increase in YOY abundance not only with increased total marsh area but also with increased island shallow salt marsh area. Just as previous investigations (Kneib 1987; Kneib and Wagner 1994) suggest that low marsh refuge during low tide is important for reduction of predation risk for YOY F. heteroclitus from predator species that utilize adjacent marsh creeks, so to is shallow salt marsh habitat important during the high tide stage.

Intraspecific predation on Fundulus heteroclitus YOY might also have been a strong contributor towards explaining differences among sites for YOY. While intraspecific predation would typically be considered density dependant and populations at islands were relatively low, thresholds for such interactions could be reduced at island salt marshes since isolation and limited refuge habitat may increase encounter rates. Similar mortality might have occurred at interior salt marsh due to high population abundance (Chapter 2). Fringing mainland salt marsh had intermediate adult abundance and connection to like habitat reducing potential adult-recruit interactions. Cannibalism by F. heteroclitus on its young, which has been suggested to be important in the distribution of YOY F. heteroclitus within mainland creek salt marshes (Kneib 1987; Halpin 2000), also might have significantly influenced population maintenance and structure at the salt marshes examined, particularly within isolated island salt marshes.

Further investigations might provide additional information that could shed light on why the differences for juvenile and adult Fundulus heteroclitus abundance patterns

occurred among the various sized salt marsh types. However, regardless of causality, it was evident that not only does the size of salt marsh habitat influence its ability to maintain salt marsh-dependant nekton, but the size of sub-habitats, particularly those important for predation refuge, can significantly influence population maintenance. The degree of isolation from like habitat, location of the salt marsh with respect to open water areas and habitat area/perimeter attributes also affect large nekton usage which can lead to higher predation impacts on smaller resident salt marsh species. It was evident that island salt marsh habitat not only provided different functional support for nekton, but the potential for trophic energy exchange through predation from salt marsh-dependant to mobile nekton predators was greater within the island salt marshes compared to mainland and interior salt marsh habitat. It is possible to manage salt marsh habitat for different nekton production and support strategies. Interior and mainland salt marshes can be managed to provide salt marsh-dependant nekton, while large island salt marshes provide enhanced support for more open water pelagic oriented nekton and nekton predator species. To best use resources to responsibly manage estuaries the functional effects of habitat attributes and their impact on nekton support dynamics need to be considered when habitat preservation and restoration efforts are considered.

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Table 3-1. Mean annual physical parameters measured for the salt marsh types. Significant differences ($p \leq 0.05$) between marsh types are designated by different letters. Mainland salt marsh areas are minimal estimates based on the smallest mainland salt marsh site. Data that had to be nlog + 1 transformed prior to analysis to meet homogeneity assumptions are designated by a single asterisk. Data that did not meet homogeneity assumption after transformation and were analyzed using a non-parametric test are designated by two asterisks.

| Marsh Type | Marsh Fringe Elevation (cm) | Distance Water Intrusion | Marsh Slope | Vertical Relief (cm) | Total Marsh* Size ($m^2$) | Low Marsh* Size ($m^2$) | Shallow* Marsh Size ($m^2$) | Distance** to Nearest Marsh | Salinity (ppt) | Water Temperature ($^oC$) |
|---|---|---|---|---|---|---|---|---|---|---|
| | ---------- Year One ---------- | | | | | | | | | |
| Mainland | -78.5 | 22.9 | 0.02753 | $45.2^{B}$ | $>76,000^{A}$ | $>19,300^{A}$ | $>56,700^{A}$ | $0^{B}$ | 32.3 | 18.1 |
| Large Island | -81.5 | 25.8 | 0.02912 | $87.6^{A}$ | $5,605^{B}$ | $2,800^{B}$ | $2,805^{B}$ | $705^{A}$ | 32.0 | 19.0 |
| Small Island | -94.9 | 15.1 | 0.04008 | $73.1^{A}$ | $629^{C}$ | $582^{C}$ | $47^{C}$ | $439^{A}$ | 33.7 | 17.9 |
| | ---------- Year Two ---------- | | | | | | | | | |
| Mainland | -68.4 | 20.6 | 0.03014 | $45.2^{B}$ | $>76,000^{A}$ | $>19,300^{A}$ | $>56,700^{A}$ | $0^{B}$ | 32.4 | $17.6^{AB}$ |
| Large Island | -72.6 | 25.1 | 0.03283 | $87.6^{A}$ | $5,605^{B}$ | $2,800^{B}$ | $2,805^{B}$ | $705^{A}$ | 33.8 | $16.2^{B}$ |
| Small Island | -80.3 | 13.3 | 0.04969 | $73.1^{A}$ | $629^{C}$ | $582^{C}$ | $47^{C}$ | $439^{A}$ | 31.3 | $19.7^{A}$ |

Table 3-2. Mean number of stems 1 $m^{-2}$ per date and for each year as a whole. For each species during each collection date, comparisons between mainland marsh (MLM), large island marsh (LIM) and small island marsh (SIM) treatments that are significantly different from one another at $p \leq 0.05$ are indicated by a different letter. Numbers in parentheses are one standard error, L = live stems, D = dead stems, T = total of all stems.

| | June 2002 | | | November 2002 | | |
|---|---|---|---|---|---|---|
| Species/Type | MLM | LIM | SIM | MLM | LIM | SIM |
| Spartina alterniflora (L) | 195.50(±38.77) | 177.00(±49.34) | 198.67(±9.60) | 116.17(±22.65) | 39.00(±5.00) | 142.33(±36.04) |
| Spartina alterniflora (D) | 147.17(±24.92) | 83.67(±8.45) | 62.33(±27.70) | 229.50(±33.71) | 163.33(±13.53) | 212.00(±46.06) |
| Spartina alterniflora (T) | 342.67(±61.34) | 260.67(±43.18) | 261.00(±23.18) | 345.67(±54.48) | 202.33(±12.71) | 354.33(±69.82) |
| Salicornia virginica (T) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 20.16(±12.79) | 0.00(±0.00) | 0.00(±0.00) |
| Total stems all species | 342.67(±61.34) | 260.67(±43.18) | 261.00(±23.18) | 365.83(±56.57) | 202.33(±12.71) | 354.33(±69.82) |

| | March 2003 | | | Total Year 1 | | |
|---|---|---|---|---|---|---|
| | MLM | LIM | SIM | MLM | LIM | SIM |
| Spartina alterniflora (L) | 305.67(±26.10) | 203.67(±23.67) | 380.00(±87.52) | 205.78(±23.99) | 139.89(±23.57) | 240.33(±41.91) |
| Spartina alterniflora (D) | 182.00(±24.46) | 107.00(±9.02) | 152.00(±15.52) | 186.22(±24.65) | 118.00(±3.83) | 142.11(±27.91) |
| Spartina alterniflora (T) | 487.67(±32.54)$^{A}$ | 310.67(±29.49)$^{B}$ | 532.00(±90.56)$^{A}$ | 392.00(±46.56) | 257.89(±19.93) | 382.44(±59.71) |
| Salicornia virginica (T) | 14.83(±10.53) | 0.00(±0.00) | 0.00(±0.00) | 11.67(±7.62) | 0.00(±0.00) | 0.00(±0.00) |
| Total stems all species | 503.17(±29.52)$^{A}$ | 310.67(±29.49)$^{B}$ | 532.00(±90.56)$^{A}$ | 403.89(±45.27) | 257.89(±19.93) | 382.44(±59.71) |

Table 3-2. (Continued).

| Species/Type | June 2003 MLM | June 2003 LIM | June 2003 SIM | November 2003 MLM | November 2003 LIM | November 2003 SIM |
|---|---|---|---|---|---|---|
| Spartina alterniflora (L) | 178.33(±31.10) | 101.00(±11.27) | 279.33(±35.32) | 252.60(±65.18) | 106.67(±19.92) | 269.00(±54.52) |
| Spartina alterniflora (D) | 107.00(±16.00)[A] | 80.33(±6.56)[A] | 30.33(±5.24)[B] | 113.80(±27.75) | 111.33(±30.22) | 68.33(±23.78) |
| Spartina alterniflora (T) | 285.33(±43.34) | 181.33(±8.41) | 309.67(±37.60) | 366.40(±70.07) | 218.00(±20.21) | 337.33(±31.18) |
| Salicornia virginica (T) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 1.80(±1.80) | 0.00(±0.00) | 0.00(±0.00) |
| Total stems all species | 285.33(±43.34) | 181.33(±8.41) | 309.67(±37.60) | 368.20(±70.86) | 218.00(±20.21) | 337.33(±31.18) |

| | March 2004 MLM | March 2004 LIM | March 2004 SIM | Total Year 2 MLM | Total Year 2 LIM | Total Year 2 SIM |
|---|---|---|---|---|---|---|
| Spartina alterniflora (L) | 273.67(±29.73) | 224.33(±67.86) | 169.33(±20.33) | 233.94(±33.59) | 144.00(±17.13) | 239.22(±22.62) |
| Spartina alterniflora (D) | 223.33(±43.16) | 113.67(±16.48) | 163.67(±21.73) | 151.17(±24.34) | 101.78(±16.06) | 87.44(±12.09) |
| Spartina alterniflora (T) | 497.00(±72.08) | 338.00(±82.72) | 333.00(±25.06) | 385.11(±54.72) | 245.78(±33.10) | 326.67(±20.72) |
| Salicornia virginica (T) | 3.83(±3.83) | 0.67(±0.67) | 0.00(±0.00) | 1.78(±1.78) | 0.22(±0.22) | 0.00(±0.00) |
| Total stems all species | 500.83(±74.06) | 338.67(±82.95) | 333.00(±25.06) | 386.89(±55.67) | 246.00(±33.20) | 326.67(±20.72) |

Table 3-3. Total number of all nekton species by time period and for the year, collected for each marsh type via gill net collections.

| | Date | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Treatment Type | 6/02 | 11/02 | 3/03 | Total Year One | 6/03 | 11/03 | 3/04 | Total Year Two |
| Mainland Marsh | 9 | 3 | 4 | 13 | 8 | 3 | 3 | 10 |
| Large Island Marsh | 8 | 1 | 6 | 12 | 9 | 0 | 5 | 11 |
| Small Island Marsh | 7 | 4 | 4 | 10 | 9 | 2 | 0 | 10 |

Table 3-4. Number of individuals from gill net collections, catch per unit effort (one tidal cycle), by collection period, and cumulative annual totals. During each collection date, per species, comparisons between mainland marsh (MLM), large island (LIM), and small island marshes (SIM) that differ significantly are indicated by a different letter ($p \leq 0.05$).

| | 6/02 | | | 11/02 | | | 3/03 | | | Total Year One | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | MLM | LIM | SIM | MLM | LIM | SIM | MLM | LIM | SIM | MLM | LIM | SIM |
| Anchoa hepsetus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Brevoortia tyrannus | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 | 0.67 | 0.17 | 0.33 | 3.33 | 0.17 | 1.00 | 4.00 |
| Cynoscion regalis | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| Dasyatis americana | 0.17 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.33 | 0.00 |
| Elops saurus | $0.00^{B}$ | $0.00^{B}$ | $0.67^{A}$ | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | $0.00^{B}$ | $0.00^{B}$ | $0.67^{A}$ |
| Gymnura micrura | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Lagodon rhomboides | 0.50 | 0.33 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.50 | 0.33 | 0.33 |
| Leiostomus xanthurus | 0.00 | 0.33 | 0.67 | 0.00 | 0.00 | 0.33 | 0.00 | 0.67 | 0.67 | $0.00^{B}$ | $1.00^{AB}$ | $1.67^{A}$ |
| Mugil cephalus | 1.00 | 1.67 | 0.67 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 1.33 | 1.67 | 1.00 |
| Micropogon undulatus | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| Mustelus canis | 0.17 | 0.00 | 3.67 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 3.67 |
| Opsanus tau | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| Orthopristis chrysoptera | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 | 0.33 |
| Pogonias cromis | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Paralichthys albigutta | 0.17 | 1.33 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 | 0.00 | $0.17^{B}$ | $2.00^{A}$ | $0.00^{B}$ |
| Paralichthys dentatus | 0.50 | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.50 | 0.00 | 0.67 |
| Paralichthys lethostigma | 0.17 | 0.33 | 1.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.17 | 0.67 | 1.00 |
| Peprilus triacanthus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Pomatomus saltatrix | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.33 | 0.33 | 0.50 | 0.33 | 0.33 |
| Sciaenops ocellatus | 0.00 | 0.00 | 0.00 | 0.50 | 0.00 | 0.67 | 0.83 | 0.00 | 0.00 | 1.33 | 0.00 | 0.67 |
| Sphyrna tiburo | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Squalus acanthias | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 | 0.67 | 0.00 |
| Strongylura notata | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.33 | 0.00 |
| Synodus foetens | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

Table 3-4. Continued..

| Species | 6/03 MLM | 6/03 LIM | 6/03 SIM | 11/03 MLM | 11/03 LIM | 11/03 SIM | 3/04 MLM | 3/04 LIM | 3/04 SIM | Total Year Two MLM | Total Year Two LIM | Total Year Two SIM |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Anchoa hepsetus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.50 | 0.00 | 0.00 | 0.50 | 0.00 | 0.00 |
| Brevoortia tyrannus | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.33 | 1.17 | 21.67 | 0.00 | 1.17 | 22.00 | 0.33 |
| Cynoscion regalis | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Dasyatis americana | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Elops saurus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Gymnura micrura | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| Lagodon rhomboides | 0.50 | 1.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.50 | 1.00 | 0.33 |
| Leiostomus xanthurus | 0.50 | 0.33 | 0.67 | 0.00 | 0.00 | 0.00 | $0.17^{B}$ | $1.00^{A}$ | $0.00^{B}$ | 0.67 | 1.33 | 0.67 |
| Mugil cephalus | 1.50 | 1.67 | 1.00 | 0.17 | 0.00 | 0.67 | 0.00 | 2.00 | 0.00 | 1.67 | 3.67 | 1.67 |
| Micropogon undulatus | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 |
| Mustelus canis | 0.00 | 1.33 | 11.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.33 | 11.00 |
| Opsanus tau | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Orthopristis chrysoptera | 0.17 | 0.67 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.67 | 0.00 |
| Pogonias cromis | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Paralichthys albigutta | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| Paralichthys dentatus | 0.17 | 0.33 | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.33 | 0.00 |
| Paralichthys lethostigma | 1.17 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.17 | 0.33 | 0.00 |
| Peprilus triacanthus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 8.67 | 0.00 | 0.00 | 8.67 | 0.00 |
| Pomatomus saltatrix | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 1.67 | 0.00 | 0.00 | 1.67 | 0.33 |
| Sciaenops ocellatus | 0.00 | 0.33 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.33 |
| Sphyrna tiburo | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 |
| Squalus acanthias | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Strongylura notata | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Synodus foetens | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |

Table 3-5. Number of individuals from gill net collections, catch per unit effort (during one complete tidal cycle), for each marsh type by collection period for years one and two. For each species during each collection date, comparisons between collection period per year for each marsh type that are significantly different from one another are indicated by a different letter ($p \leq 0.05$).

| | ------Mainland------ | | | ----Large Island---- | | | ----Small Island --- | | |
|---|---|---|---|---|---|---|---|---|---|
| Species | 6/02 | 11/02 | 3/03 | 6/02 | 11/02 | 3/03 | 6/02 | 11/02 | 3/03 |
| Anchoa hepsetus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Brevoortia tyrannus | 0.00 | 0.00 | 0.17 | 0.67 | 0.00 | 0.33 | 0.00 | 0.67 | 3.33 |
| Cynoscion regalis | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Dasyatis americana | 0.17 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Elops saurus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 |
| Gymnura micrura | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Lagodon rhomboides | $0.50^{A}$ | $0.00^{B}$ | $0.00^{B}$ | 0.33 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 |
| Leiostomus xanthurus | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.67 | 0.67 | 0.33 | 0.67 |
| Mugil cephalus | 1.00 | 0.33 | 0.00 | $1.67^{A}$ | $0.00^{B}$ | $0.00^{B}$ | 0.67 | 0.00 | 0.33 |
| Micropogon undulatus | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Mustelus canis | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 3.67 | 0.00 | 0.00 |
| Opsanus tau | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Orthopristis chrysoptera | 0.00 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 |
| Pogonias cromis | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Paralichthys albigutta | 0.17 | 0.00 | 0.00 | 1.33 | 0.67 | 0.00 | 0.00 | 0.00 | 0.00 |
| Paralichthys dentatus | 0.50 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 |
| Paralichthys lethostigma | 0.17 | 0.00 | 0.00 | 0.33 | 0.00 | 0.33 | 1.00 | 0.00 | 0.00 |
| Peprilus triacanthus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Pomatomus saltatrix | 0.33 | 0.00 | 0.17 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.33 |
| Sciaenops ocellatus | 0.00 | 0.50 | 0.83 | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 | 0.00 |
| Sphyrna tiburo | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Squalus acanthias | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 |
| Strongylura notata | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 |
| Synodus foetens | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

Table 3-5. Continued.

| | ------Mainland------ | | | ----Large Island---- | | | ----Small Island --- | | |
|---|---|---|---|---|---|---|---|---|---|
| Species | 6/03 | 11/03 | 3/04 | 6/03 | 11/03 | 3/04 | 6/03 | 11/03 | 3/04 |
| Anchoa hepsetus | 0.00 | 0.00 | 0.50 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 |
| Brevoortia tyrannus | 0.00 | 0.00 | 1.17 | 0.33 | 0.00 | 21.67 | 0.00 | 0.00 | 0.00 |
| Cynoscion regalis | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Dasyatis americana | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Elops saurus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Gymnura micrura | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Lagodon rhomboides | 0.50 | 0.00 | 0.00 | 1.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 |
| Leiostomus xanthurus | 0.50 | 0.00 | 0.17 | 0.33 | 0.00 | 1.00 | 0.67 | 0.00 | 0.00 |
| Mugil cephalus | 1.50 | 0.17 | 0.00 | 1.67 | 0.00 | 2.00 | 1.00 | 0.67 | 0.00 |
| Micropogon undulatus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 |
| Mustelus canis | 0.00 | 0.00 | 0.00 | 1.33 | 0.00 | 0.00 | 11.00 | 0.00 | 0.00 |
| Opsanus tau | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Orthopristis chrysoptera | 0.17 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Pogonias cromis | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 |
| Paralichthys albigutta | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Paralichthys dentatus | 0.17 | 0.17 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Paralichthys lethostigma | 1.17[A] | 0.00[B] | 0.00[B] | 0.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Peprilus triacanthus | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 8.67 | 0.00 | 0.00 | 0.00 |
| Pomatomus saltatrix | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1.67 | 0.33 | 0.00 | 0.00 |
| Sciaenops ocellatus | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 |
| Sphyrna tiburo | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 |
| Squalus acanthias | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Strongylura notata | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Synodus foetens | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

Table 3-6. Area-perimeter and area-perimeter-predator-concentration ratios for the salt marsh types. Significant differences ($p \leq 0.05$) between marsh types are designated by different letters. Mainland salt marsh areas and predation perimeters are estimated based on ecological boundaries at each site and predator concentrations were based on cumulative predator catch per site. Data was nlog + 1 transformed prior to analysis to meet homogeneity assumptions.

| Marsh Type | Low Marsh Area/Perimeter | Area/Perimeter Predator |
|---|---|---|
| Mainland | $20.6^{A}$ | $3.6^{A}$ |
| Large Island | $9.8^{AB}$ | $0.5^{B}$ |
| Small Island | $4.6^{B}$ | $0.3^{B}$ |

Fig. 3-1. Site locations within Bogue, Back and Core Sounds of North Carolina. Locations corresponding to the circles are mainland salt marsh sites, squares are large island salt marsh sites, stars are small island salt marsh sites and triangles are interior salt marsh sites.

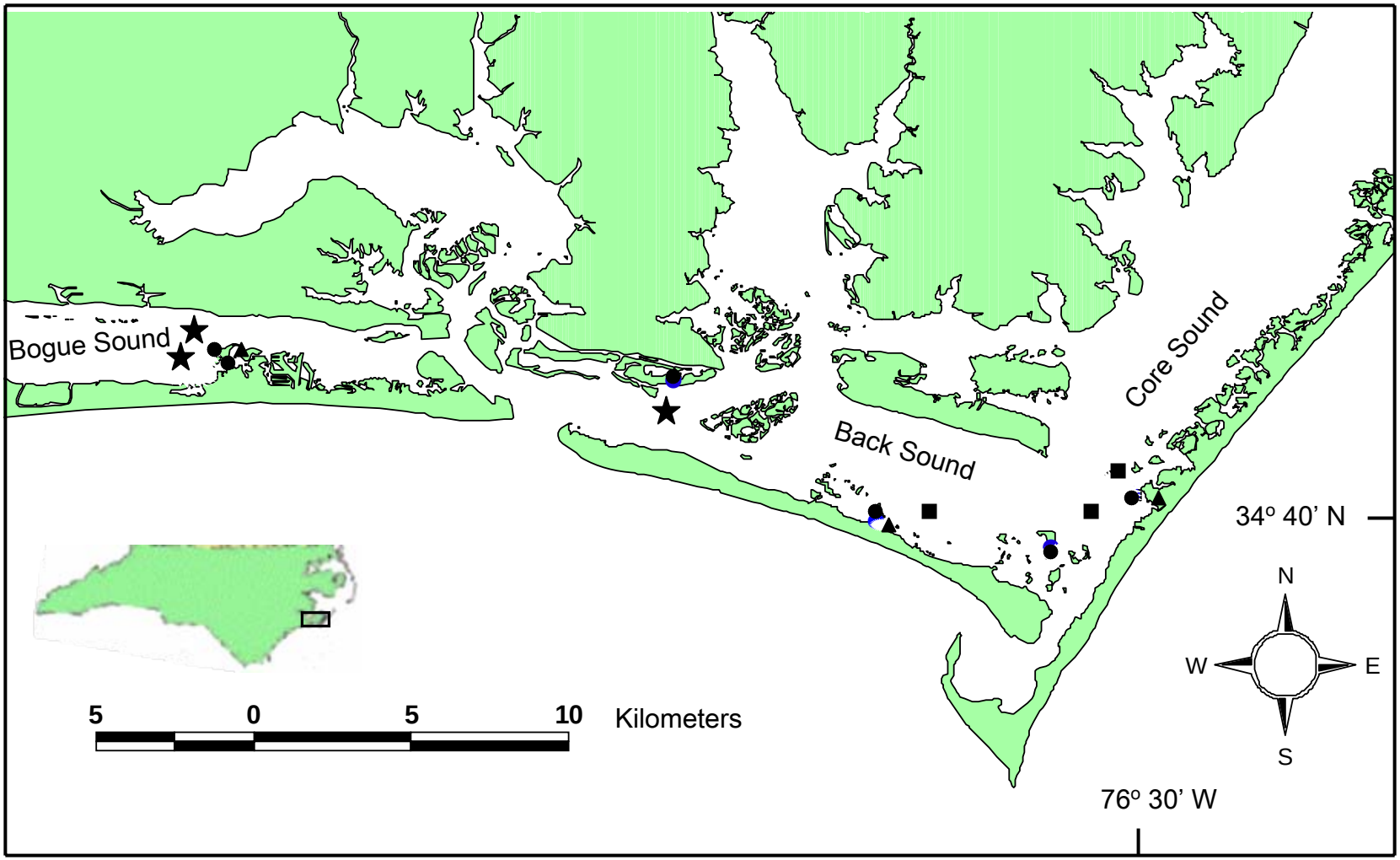
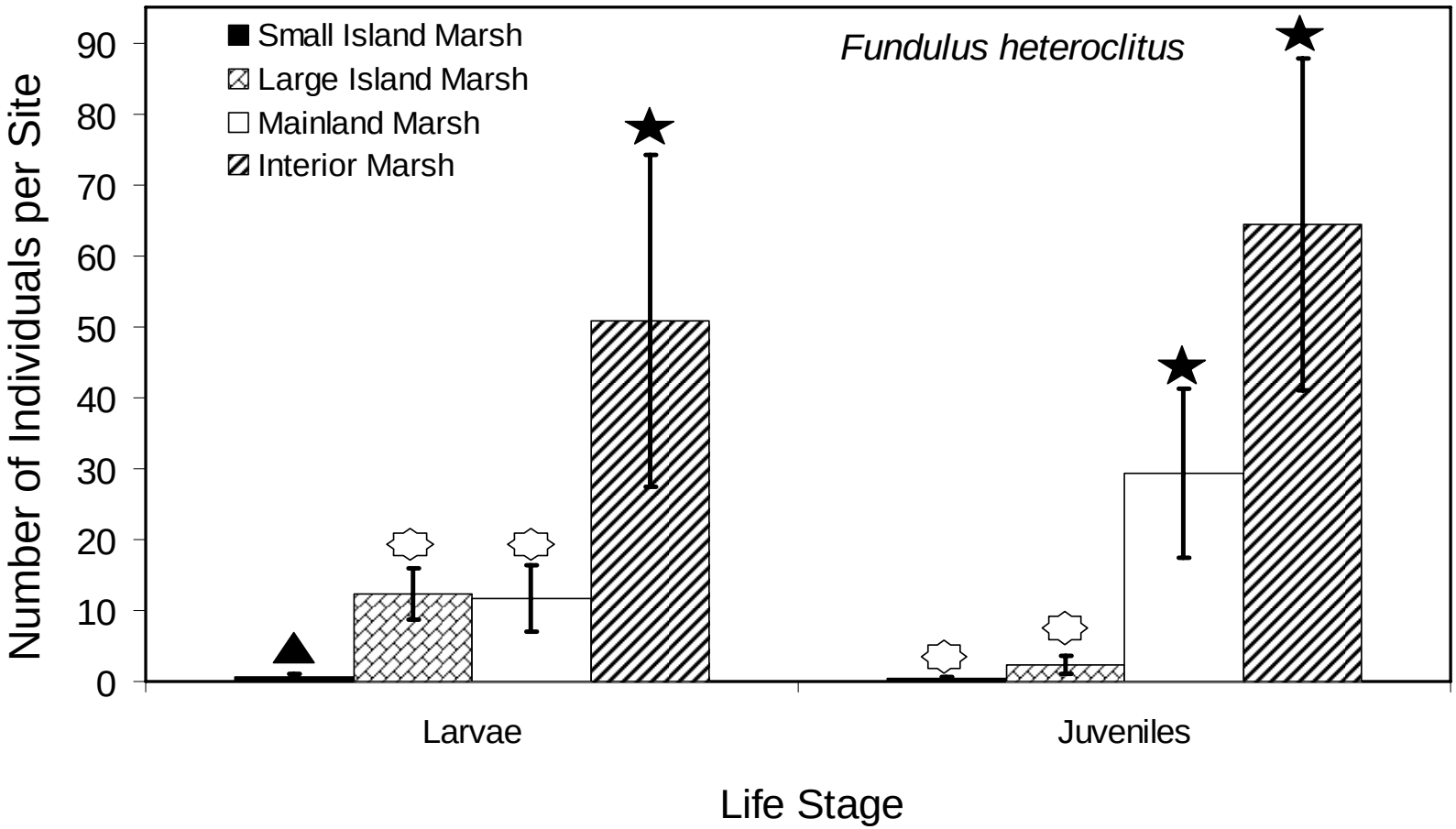
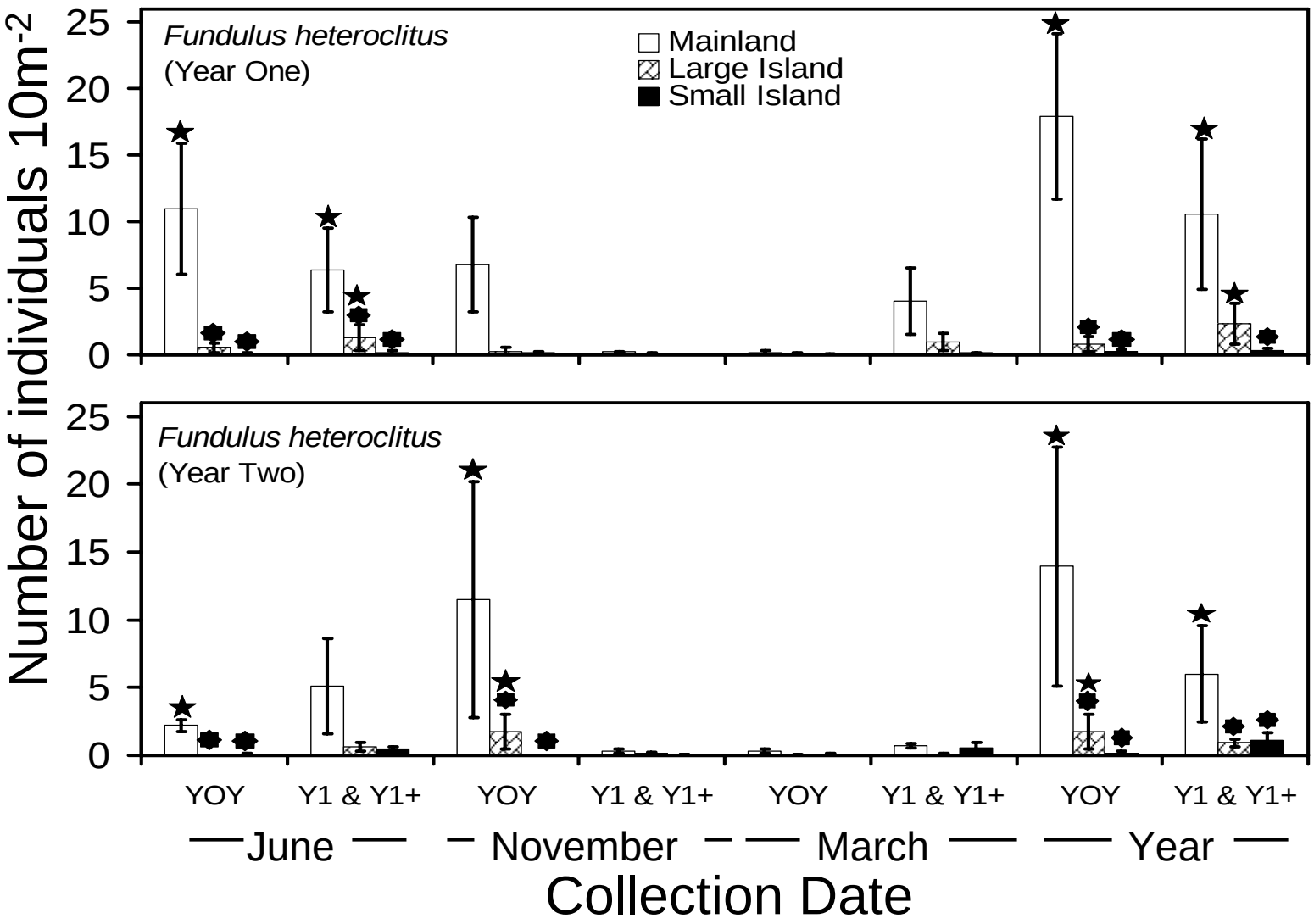

Bogue Sound
Back Sound
Core Sound
34° 40' N
76° 30' W
N
W
E
S
5
0
5
10
Kilometers

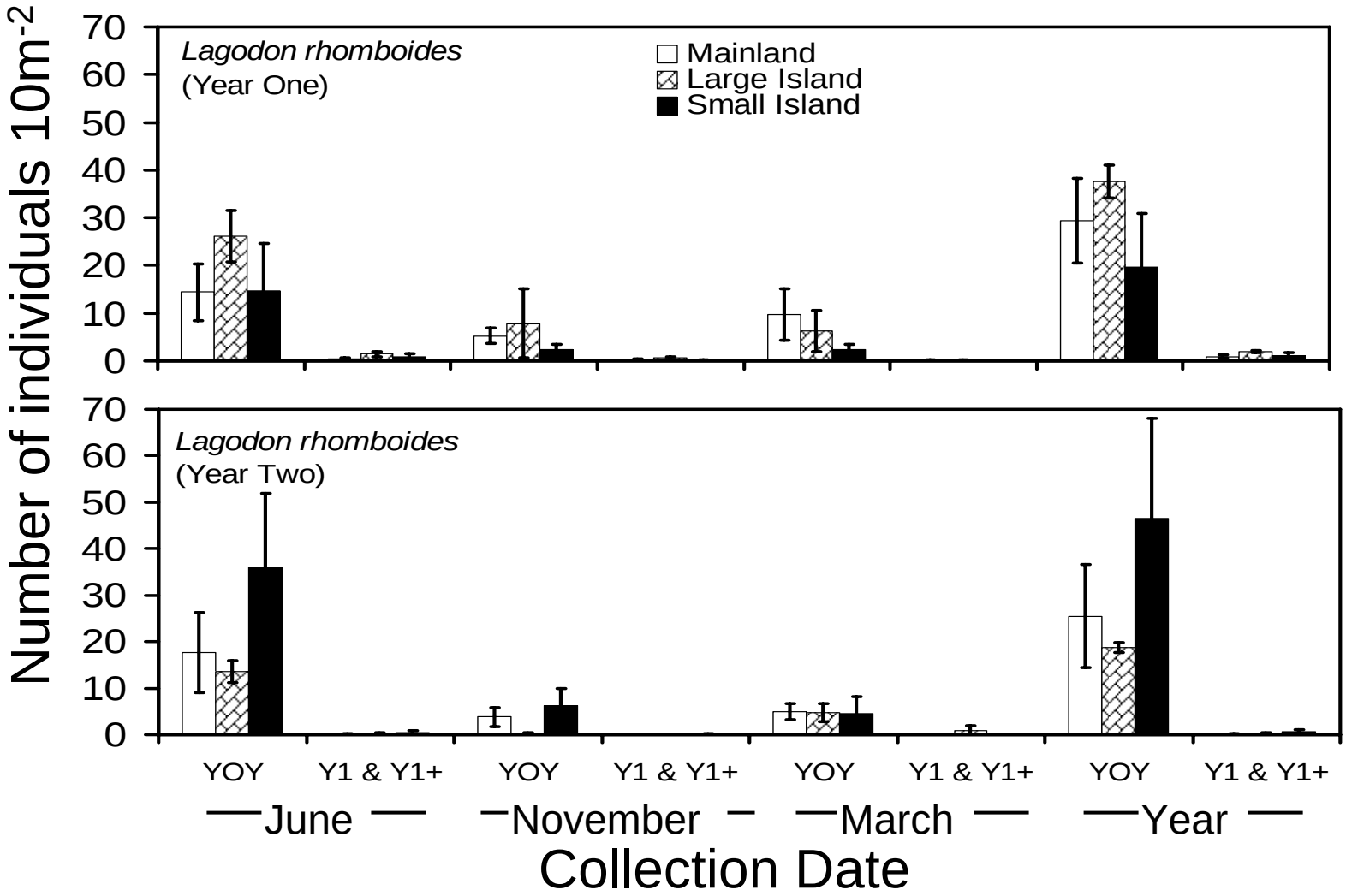
Fig. 3-2. Mean number of Fundulus heteroclitus larvae and juveniles collected per salt marsh type during Breder and pit trap collections. For each life history stage, salt marsh mean comparisons among salt marsh types that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. One standard error is indicated by the error bars.


Small Island Marsh
Large Island Marsh
Mainland Marsh
Interior Marsh
Fundulus heteroclitus
Number of Individuals per Site
0
10
20
30
40
50
60
70
80
90
Larvae
Juveniles
Life Stage

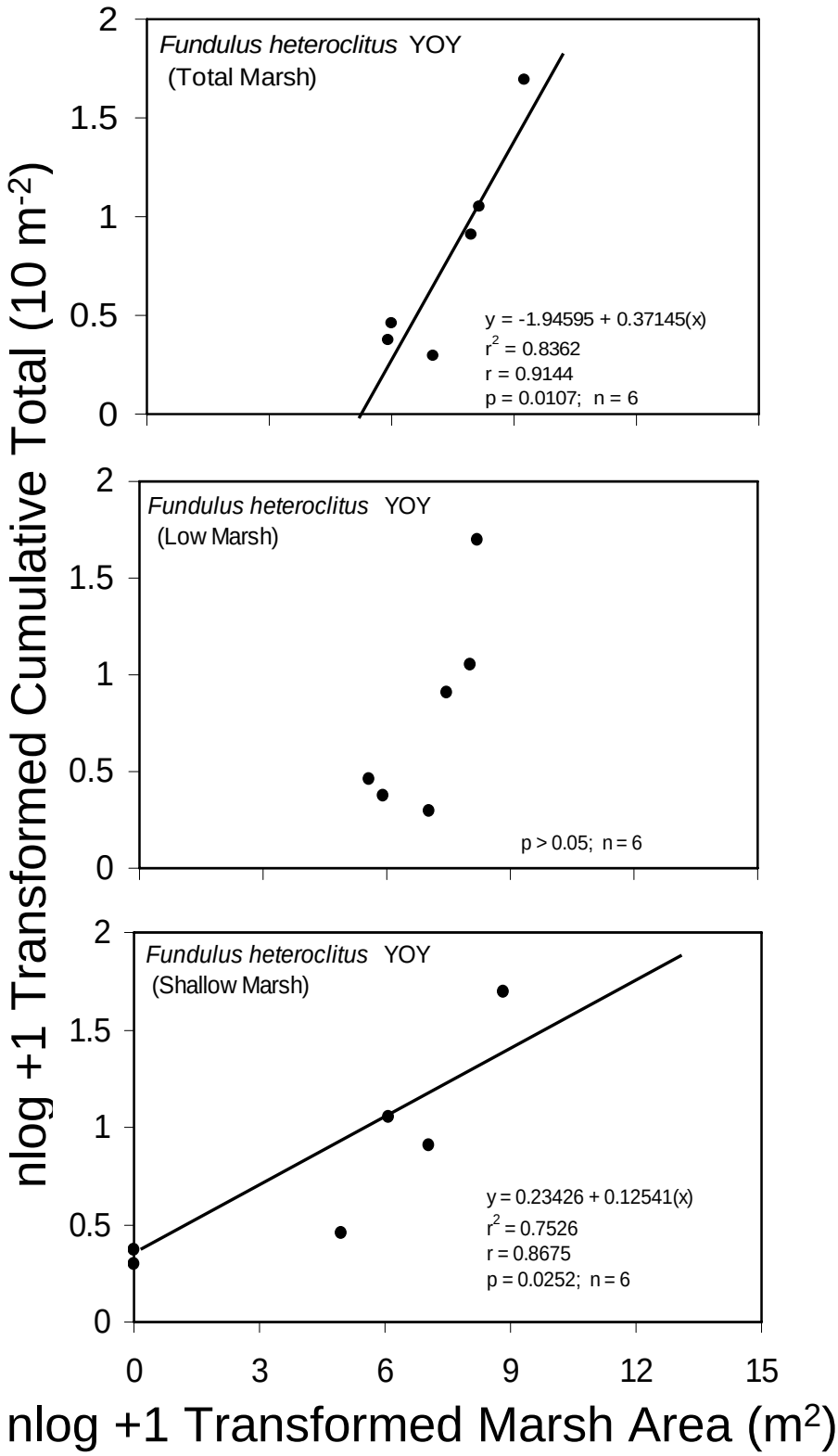
Figs. 3-3a,b. Mean number of Fundulus heteroclitus young of year and year one, year one + cohorts collected per salt marsh type, per collection date and annual cumulative catch per year, for years one and two during fyke and block net collections. For each cohort, collection date and annual cumulative catch, mean comparisons among salt marsh types that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. One standard error is indicated by the error bars.


Fundulus heteroclitus
(Year One)
Mainland
Large Island
Small Island
Fundulus heteroclitus
(Year Two)
Number of individuals 10m-2
0
5
10
15
20
25
YOY
Y1 & Y1+
June
November
March
Year
Collection Date

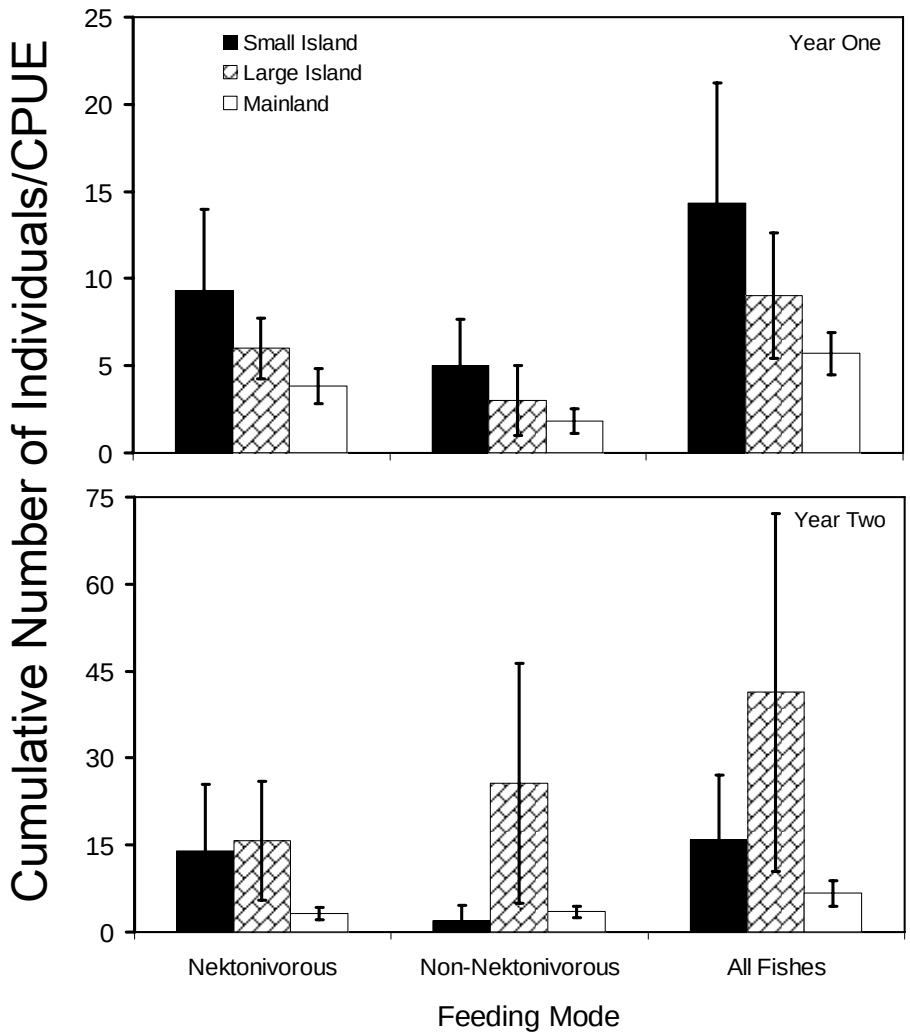
Figs. 3-4a,b. Mean number of Lagodon rhomboides young of year and year one, year one + cohorts collected per salt marsh type, per collection date and annual cumulative catch per year, for years one and two during fyke and block net collections. For each cohort, collection date and annual cumulative catch, mean comparisons among salt marsh types that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. One standard error is indicated by the error bars.


Number of individuals 10m-2
70
60
50
40
30
20
10
0
Lagodon rhomboides
(Year One)
Mainland
Large Island
Small Island
Lagodon rhomboides
(Year Two)
YOY
Y1 & Y1+
June
November
March
Year
Collection Date

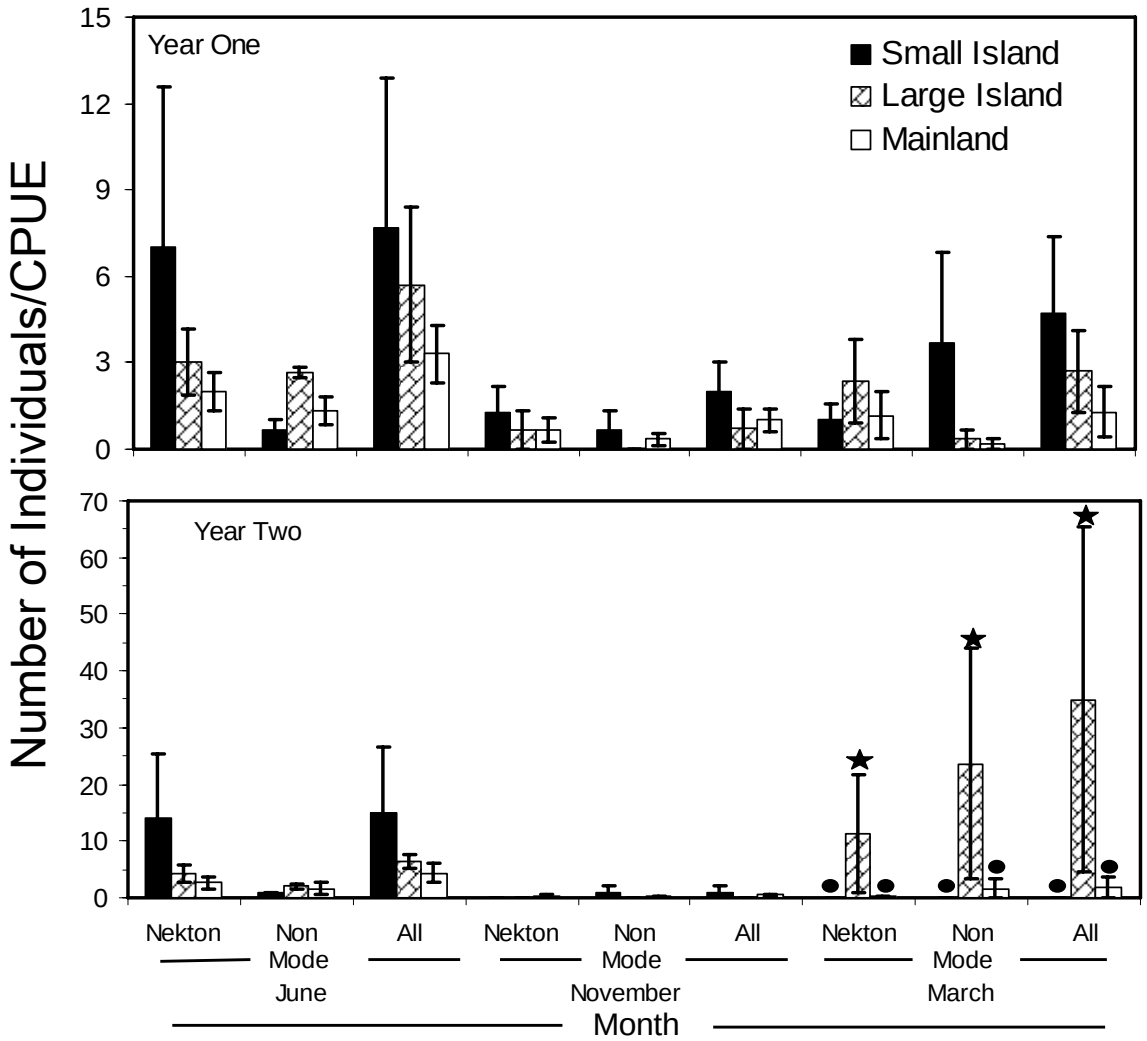
Figs. 3-5a-c. Linear regression analyses for nlog +1 transformed cumulative number of Fundulus heteroclitus young of year and year 10 $m^{-2}$ collected per island site compared to nlog +1 transformed total, low and shallow marsh areal size, $m^2$, per island. N = 6 and $p \leq 0.05$ for slope significance level.


*Fundulus heteroclitus* YOY
(Total Marsh)
y = -1.94595 + 0.37145(x)
r² = 0.8362
r = 0.9144
p = 0.0107; n = 6
*Fundulus heteroclitus* YOY
(Low Marsh)
p > 0.05; n = 6
*Fundulus heteroclitus* YOY
(Shallow Marsh)
y = 0.23426 + 0.12541(x)
r² = 0.7526
r = 0.8675
p = 0.0252; n = 6
nlog +1 Transformed Cumulative Total (10 m-2)
nlog +1 Transformed Marsh Area (m2)
0
0.5
1
1.5
2
0
3
6
9
12
15

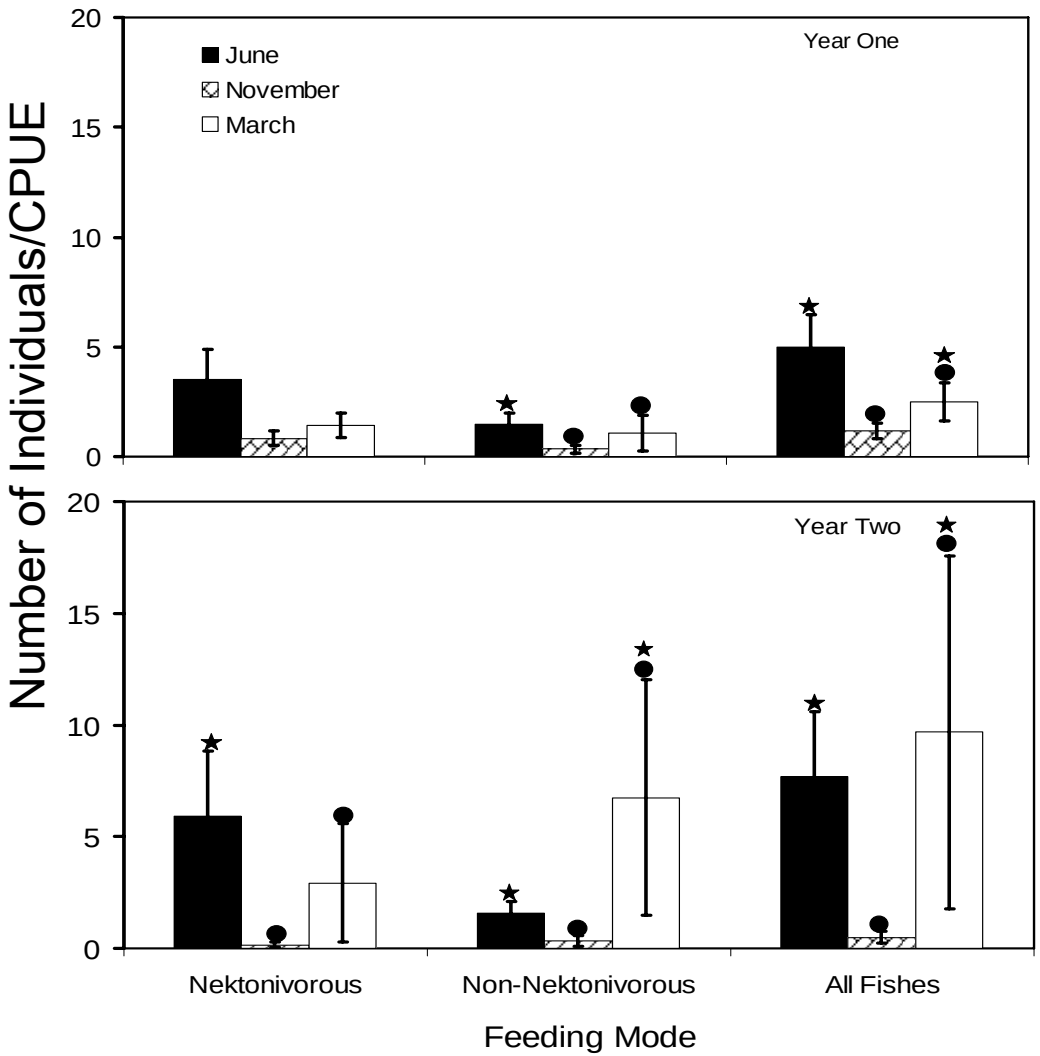
Figs. 3-6a,b. Mean annual gill net catch per unit effort (CPUE) of nektonivorous, non-nektonivorous and all fish feeding modes per marsh type for each year. For each feeding guild salt marsh types that significantly differ from one another ($p \leq 0.05$) are signified by a different symbol. Error bars correspond to one standard error.


Cumulative Number of Individuals/CPUE
Small Island
Large Island
Mainland
Year One
Year Two
25
20
15
10
5
0
75
60
45
30
15
0
Nektonivorous
Non-Nektonivorous
All Fishes
Feeding Mode

Figs. 3-7a,b. Mean gill net catch per unit effort (CPUE) of nektonivorous (nekton), non-nektonivorous (non) and all fish (all) feeding modes per salt marsh type, by collection date, for each year. For each feeding guild and collection date, salt marsh types that significantly differ from one another ($p \leq 0.05$) are signified by a different symbol. Error bars correspond to one standard error.


Year One
Small Island
Large Island
Mainland
Year Two
Number of Individuals/CPUE
15
12
9
6
3
0
70
60
50
40
30
20
10
0
Nekton
Non Mode
All
Nekton
Non Mode
All
Nekton
Non Mode
All
June
November
March
Month

Figs. 3-8a,b. Mean gill net catch per unit effort (CPUE) of nektonivorous, non-nektonivorous and all fish feeding modes, all salt marsh types combined, by collection time period for each year. For each feeding guild, collection time periods that significantly differ from one another ($p \leq 0.05$) are signified by a different symbol. Error bars correspond to one standard error.


Number of Individuals/CPUE
20
15
10
5
0
June
November
March
Year One
Year Two
Nektonivorous
Non-Nektonivorous
All Fishes
Feeding Mode

Figs. 3-9a,b. Mean gill net catch per unit effort (CPUE) of nektonivorous (nekton), non-nektonivorous (non) and all fish (all) feeding modes, per salt marsh type, by collection time period for each year. For each feeding guild, and salt marsh type, collection time periods that significantly differ from one another ($p \leq 0.05$) are signified by a different symbol. Error bars correspond to one standard error.

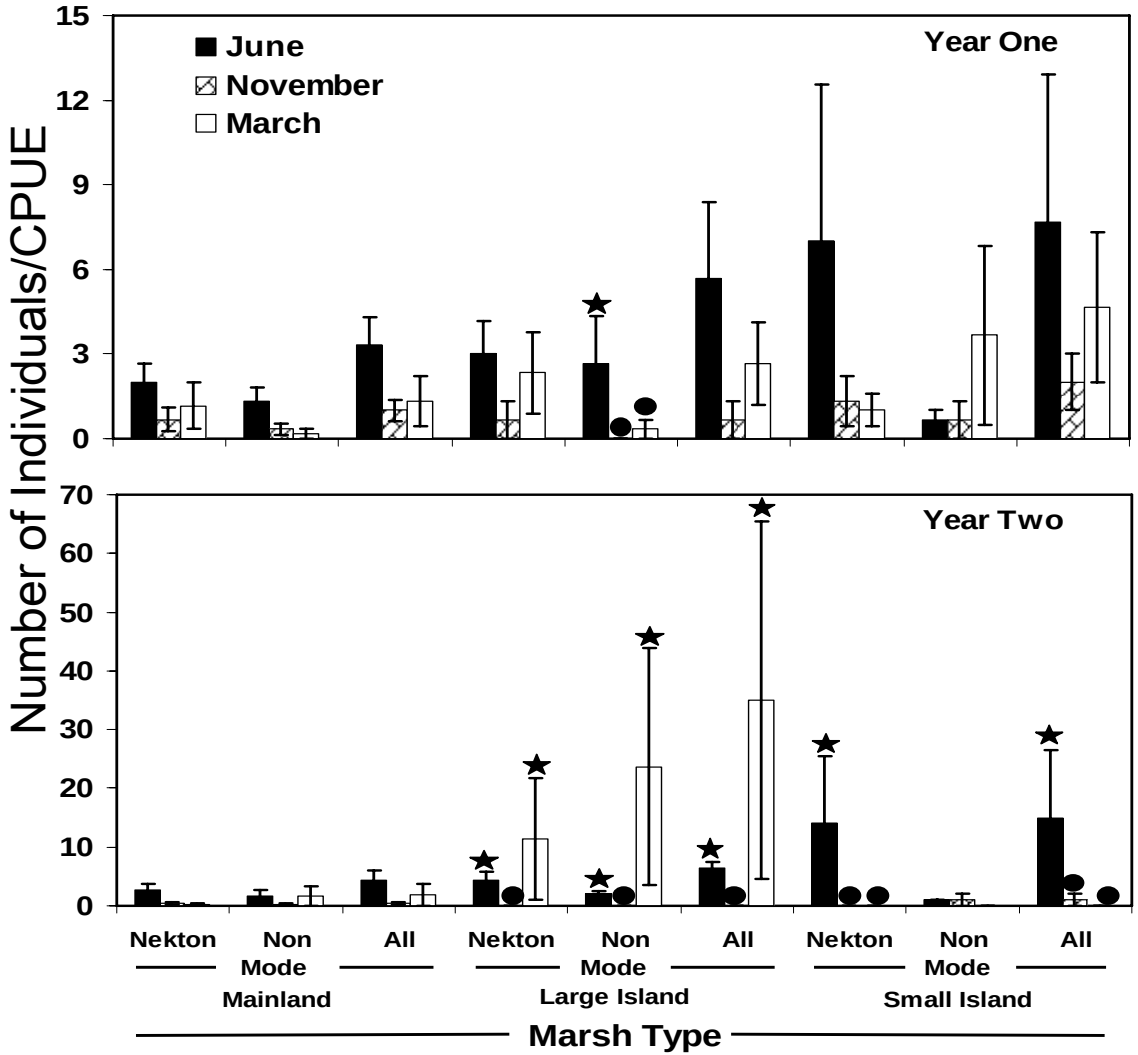
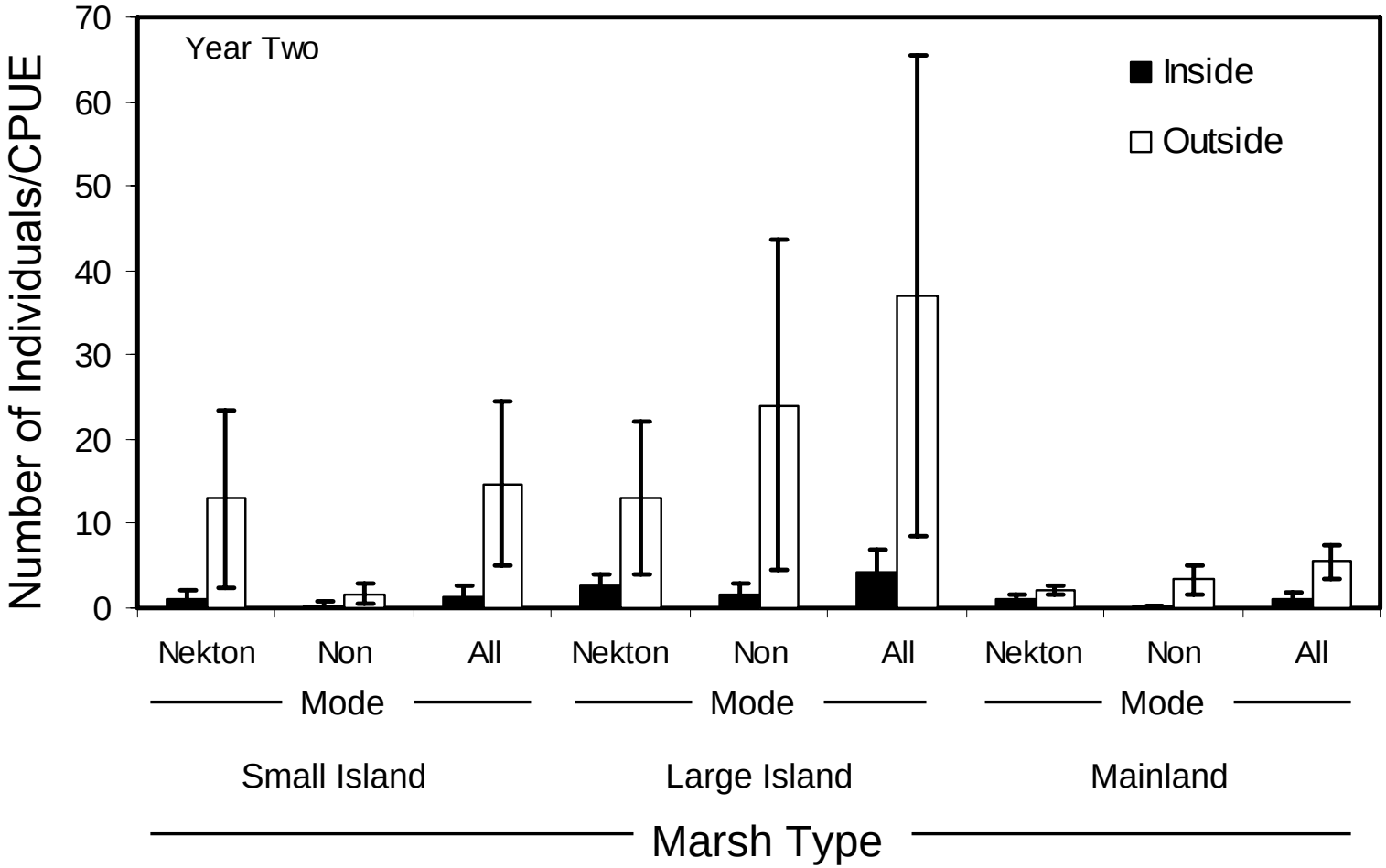
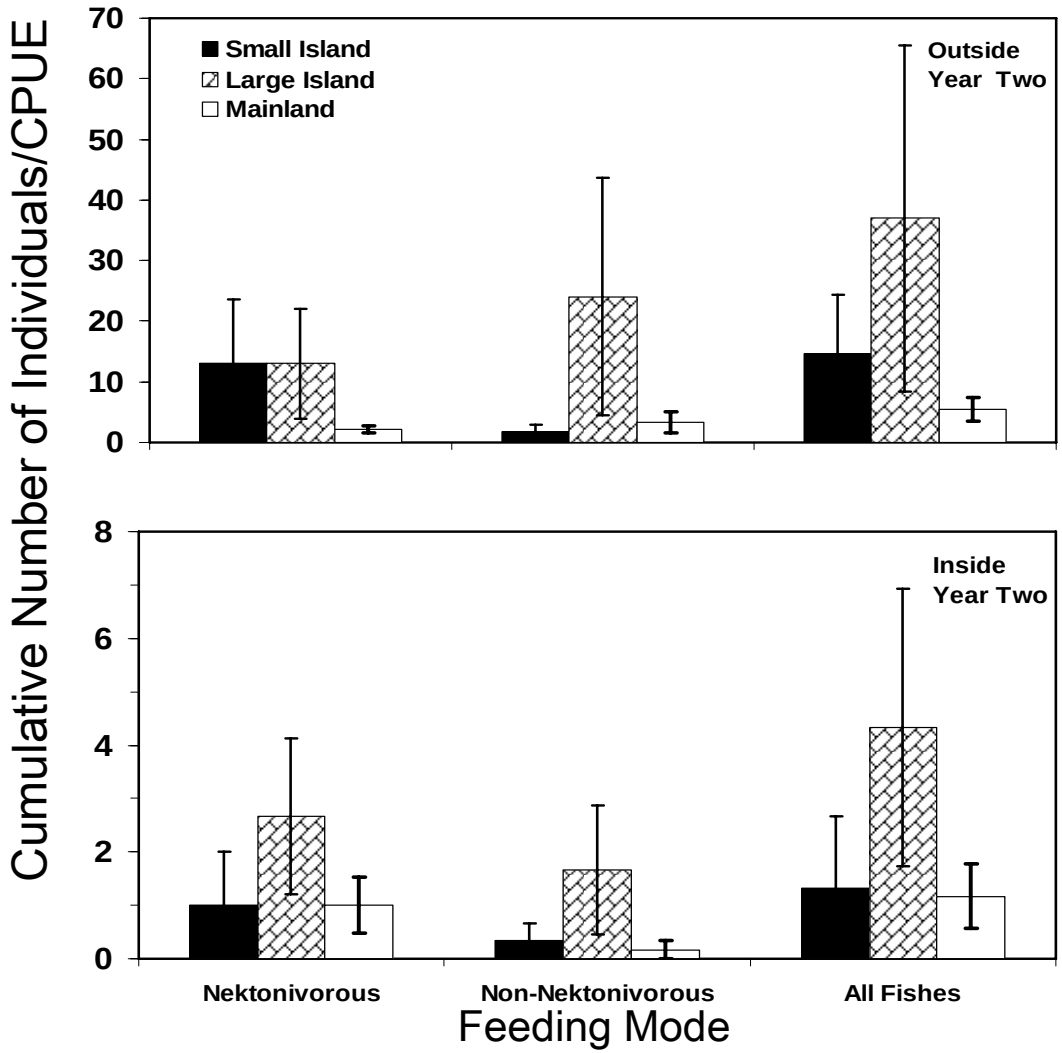

Number of Individuals/CPUE
June
November
March
Year One
Year Two
0
3
6
9
12
15
0
10
20
30
40
50
60
70
Nekton
Non Mode
All
Nekton
Non Mode
All
Nekton
Non Mode
All
Mainland
Large Island
Small Island
Marsh Type

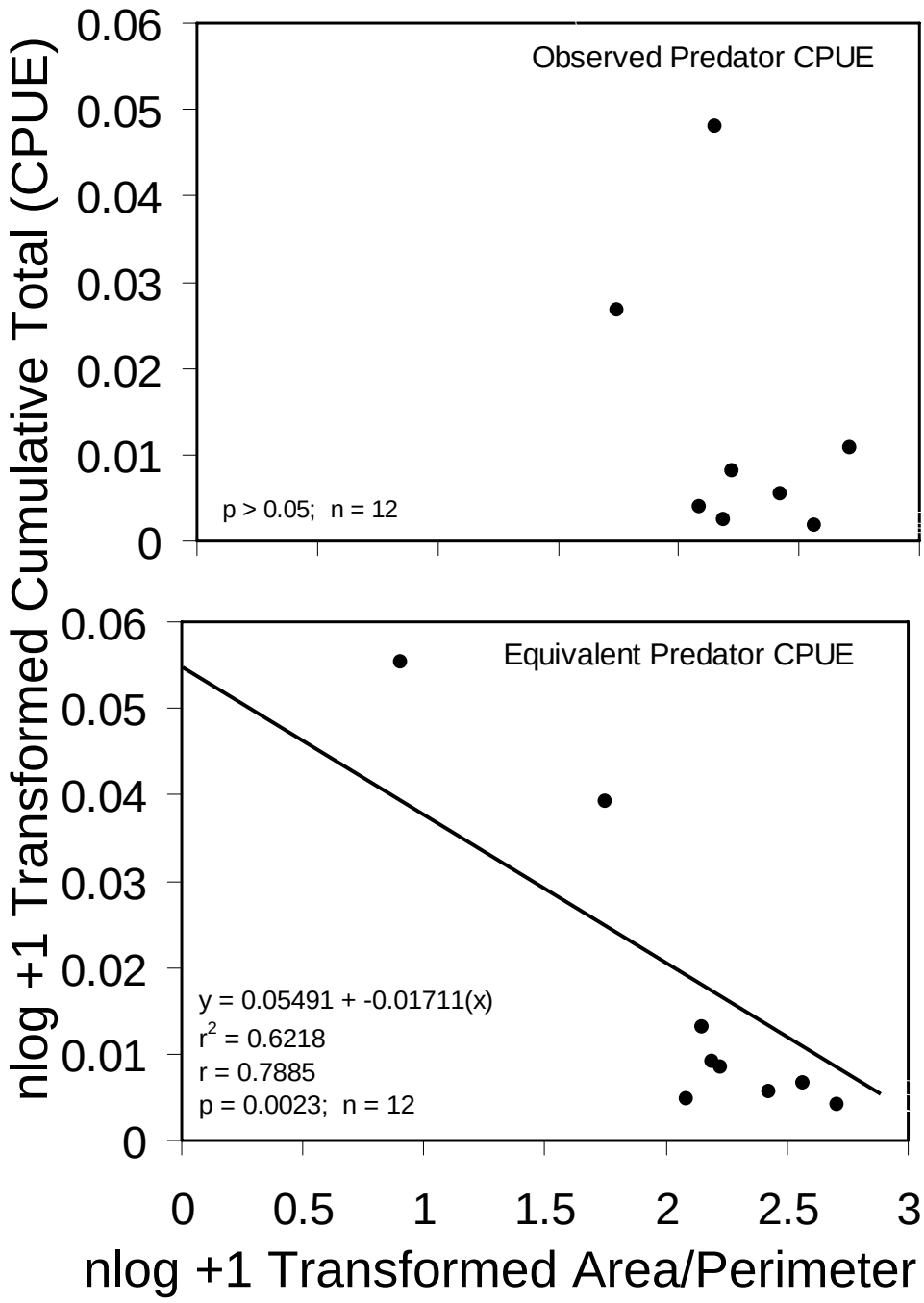
Fig. 3-10. Mean cumulative annual gill net catch per unit effort (CPUE) of nektonivorous (nekton), non-nektonivorous (non) and all fish feeding modes (all), per salt marsh type, by collection location at the salt marsh sites, inside the salt marsh (inside) and along the salt marsh fringe (outside) for year two. No significant differences ($p > 0.05$) were apparent for CPUE of the different feeding guilds for the salt marsh types between intra marsh locations.


Year Two
Inside
Outside
Number of Individuals/CPUE
70
60
50
40
30
20
10
0
Nekton
Non
All
Nekton
Non
All
Nekton
Non
All
Mode
Mode
Mode
Small Island
Large Island
Mainland
Marsh Type

Figs. 3-11a,b. Mean cumulative annual gill net catch per unit effort (CPUE) of nektonivorous, non-nektonivorous and all fish feeding modes based on collection location at the salt marsh sites, along the salt marsh fringe (outside) (a); and inside the salt marsh (inside) (b); by salt marsh type for year two. No significant differences ($p > 0.05$) were apparent for CPUE of the different feeding guilds and intra salt marsh location between salt marsh types.


Cumulative Number of Individuals/CPUE
70
60
50
40
30
20
10
0
Small Island
Large Island
Mainland
Outside
Year Two
8
6
4
2
0
Inside
Year Two
Nektonivorous
Non-Nektonivorous
All Fishes
Feeding Mode

Figs. 3-12a,b. Linear regression analyses for observed nlog +1 transformed cumulative gill net predator catch per unit effort (CPUE) per low marsh area ratios for each site (a); and equivalent nlog+1 transformed cumulative gill net predator CPUE per low marsh area ratios for each site (b); compared to nlog +1 low marsh area-low marsh perimeter edge ratios. N = 12 and $p \leq 0.05$ for slope significance level.


Observed Predator CPUE
p > 0.05; n = 12
Equivalent Predator CPUE
y = 0.05491 + -0.01711(x)
r2 = 0.6218
r = 0.7885
p = 0.0023; n = 12
nlog +1 Transformed Cumulative Total (CPUE)
nlog +1 Transformed Area/Perimeter
0
0.01
0.02
0.03
0.04
0.05
0.06
0.5
1
1.5
2
2.5
3

# CHAPTER 4.

## RE-COLONIZATION POTENTIAL OF A SALT MARSH RESIDENT, MUMMICHOG (FUNDULUS HETEROCLITUS) AND A TRANSIENT, PINFISH (LAGODON RHOMBOIDES) IN ISOLATED ISLAND SMOOTH CORDGRASS (SPARTINA ALTERNIFLORA) SALT MARSH.

# INTRODUCTION

The dispersal ability of estuarine species might directly influence their ability to populate specific estuarine habitats, especially those habitats distant from existing populations (Levin et al. 1996). While transient species that have pelagic larvae and disperse among a variety of estuarine habitats might experience less difficulty in colonizing new, or distant areas, those species that are resident to specific habitat types and have demersal, sedentary larvae might be limited in colonizing new or distant habitats (Levin et al. 1996).

Two nekton species that represent opposite dispersal ability strategies are pinfish (Lagodon rhomboides) and mummichog (Fundulus heteroclitus), which typically occur as co-dominants in salt marsh habitats along the South Atlantic coast of the USA (Hettler 1989; Meyer et al. 1996). The ability of transient estuarine species such as L. rhomboides to immigrate to and colonize salt marshes has typically not been a concern to resource managers and scientists. However, the dispersal ability and capability of dominant resident salt marsh nekton such as killifish, including F. heteroclitus, to colonize restored, created and fragmented salt marsh habitat has been a significant concern (Teo and Able 2003), due to the abundance of these fish in salt marshes (Hettler 1989; Meyer et al. 1996; 2001; Teo and Able 2003) and their role in trophic dynamics (Vince et al. 1976; Kneib 1986; 2003; Currin et al. 1995; 2003).

Fundulus heteroclitus is a key forage species for numerous ecologically and commercially important fishes (Fay et al 1983; Abrams 1985; Ruiz et al. 1993). While relatively small, rarely attaining a size larger than 100 mm standard length (SL) over the course of their four year life span (Kneib and Stiven 1978), it is known to be a voracious

predator in benthic salt marsh communities, potentially affecting benthic community composition (Vince et al. 1976; Posey and Hines 1991). The known range of F. heteroclitus is from New Hampshire to northeastern Florida, and this species may occur from hypersaline (Abrams, 1985) to freshwater conditions (Rozas and Hackney 1984; Abrams 1985; Meyer et al. 2001). Fundulus heteroclitus has an affinity for intertidal salt marsh habitats (Kneib 1984; 1986; Abrams 1985; McIvor and Odum 1986; 1988; Halpin 1997; 2000; Rozas and Zimmerman 2000) in which they spawn.

Fundulus heteroclitus spawning typically occurs from March through September (Abrams 1985), depending on water temperature (Brummett 1966), with peak spawning normally occurring by June/July (Talbot and Able 1984). Fundulus heteroclitus males become sexually mature at ~32 mm SL while females become sexually mature at ~38 mm SL (Abrams 1985). Females lay demersal eggs during high spring tides (Taylor et al. 1979) attached to benthic structures such as smooth cordgrass (Spartina alterniflora) stems (Taylor and DiMichele 1983) and shells (Able and Castanga 1975; Kneib and Stiven 1978; Taylor et al. 1981; Taylor and DiMichele 1983). A size dependant distribution of F. heteroclitus has been suggested based on predation risk (Ruiz et al. 1993), with larval F. heteroclitus typically restricted to the intertidal marsh zone, utilizing shallow marsh pools as refuges (Taylor et al. 1979 Able and Hagen 2000). Juvenile F. heteroclitus move in similar fashion to adults, and utilize the marsh surface during high tide (Kneib 1986; Rozas and Odum 1987), receding during ebb tide to shallow sublitoral habitat (Rozas and Odum 1987; Ruiz et al. 1993). While F. heteroclitus have been noted to predominantly utilize low salt marsh habitat over that of shallower high salt marsh habitats (Kneib and Wagner 1994), it is apparently restricted to the use of shallow water

regions, not exceeding ~1.0 m in depth, in unvegetated areas (Ruiz et al. 1993) and submerged aquatic vegetation (SAV) during ebb tide (Rozas and Odum 1987).

Fundulus heteroclitus has been considered to have restricted movement (< 400 m) along salt marsh creek banks (Lotrich 1975, Abrams, 1985) and within salt marsh habitat (Teo and Able 2003), and has been observed to move only 36 m laterally from summer home ranges (Lotrich 1975). Its movement and dispersal ability may be greater during nocturnal time periods (Sogard and Able 1994). Halpin (1997; 2000) noted a seasonal shift in habitat use in salt marsh habitats with increased use of mudflat by F. heteroclitus during periods when predators were less common. Teo and Able (2003) observed high site fidelity and little inter salt marsh creek movement, though substantial intra salt marsh creek movement occurred. Predators of F. heteroclitus include striped bass, (Morone saxatilis) (Fay et al. 1983), red drum, (Sciaenops ocellata), summer flounder (Paralichthys dentatus) (Abrams 1985), blue crab (Callinectes sapidus), and Atlantic croaker (Micropogonias undulatus) (Ruiz et al. 1993).

Along the South Atlantic coast of the USA, Lagodon rhomboides is ubiquitous within coastal estuarine habitats, including seagrass beds (Fonseca et al. 1990; 1996; Meyer et al. 1999; Hovel et al. 2002; Paperno et al. 2001), oyster reefs (Meyer et al. 1996) and salt marshes (Hettler 1989, Meyer et al. 1996). Lagodon rhomboides is typically a more marine oriented species than is F. heteroclitus, occurring in warm water, meso-haline estuarine and marine areas (Paperno et al. 2001), from Massachusetts though the Gulf of Mexico (Hoese et al. 1977).

In contrast to F. heteroclitus, L. rhomboides attains a larger size of up to 250 mm SL (Hoese et al. 1977), is a oceanic pelagic spawner, with larval transport into coastal

estuaries occurring during winter months along the North Carolina coast, with peak recruitment from January through March (Warlen and Burke 1990). Once within the estuaries the larvae facultatively utilize a variety of habitats and do not appear to be constrained by the lack of any one particular habitat type. Lagodon rhomboides is an important forage species for higher trophic level marine fauna (Allen et al. 2001), and also a voracious predator (Hoese et al. 1977). Feeding habits for L. rhomboides vary with ontogenetic stage, generally changing from predominantly carnivorous to omnivorous behavior with increasing size and age (Carr and Adams 1973; Stoner 1980).

The proximity of a habitat to other similar habitats, regardless of size, and size of the habitat regardless of degree of isolation from like habitats has been theorized to be important in terms of potential recruitment of new individuals and the number of species that can be supported (Rieman and McIntyre 1995; Acosta 1999; Lockwood et al. 2002). Similarly, potential colonization by species with limited dispersal ability might be low as predicted by metapopulation (Harrison and Taylor 1997; Wahlberg et al. 2002) and island biogeography theories (MacArthur and Wilson 1967; Simberloff and Wilson 1969; Simberloff and Abele 1976; Wahlberg et al. 2002), or take longer for colonization than less remote locations (MacArthur and Wilson 1967; Simberloff and Wilson 1969; Simberloff and Abele 1976). The function of habitats, regardless of size, might be temporally impacted with initial colonization lags and reduced long term re-colonization due to isolation from like habitats. If this is the case, then site proximity to like habitats is important for effective functioning of restored salt marsh habitat, and size might similarly be important for population stability.

Compared to contiguous salt marshes, the expanse of shallow and deep water

habitats that surround island salt marshes may restrict immigration and emigration of salt marsh residents due to increased predation vulnerability (Heck and Thoman 1981), thereby possibly isolating island salt marsh populations. The population size of species with poor dispersal ability that reside within isolated salt marsh islands might be more adversely affected by a local lack of resources than non-isolated habitat, and hence habitat carrying capacity might be lower. Effects may be particularly evident for created salt marshes that are initially simple in terms of habitat complexity and faunal diversity (Minello and Zimmerman 1992; Sacco et al. 1994; Levin et al. 1996).

This study used naturally occurring salt marsh islands and a mainland salt marsh site to examine dispersal and re-colonization ability of Fundulus heteroclitus and Lagodon rhomboides to isolated salt marsh islands. The objectives of this study were to: 1) measure re-colonization rate of depopulated isolated island salt marsh by F. heteroclitus and L. rhomboides, and 2) compare population size structure of F. heteroclitus and L. rhomboides at isolated salt marsh islands after removal sampling to reference mainland salt marsh.

## MATERIALS AND METHODS

### Sites

Three naturally occurring island salt marshes (~3,000-10,000 $m^2$) and a nearby naturally occurring 'mainland' salt marsh over 76,000 $m^2$ in total salt marsh size, Gunning Hammock, were used to monitor Fundulus heteroclitus and Lagodon rhomboides abundance responses to removal on the islands and non-manipulation on the mainland. Two of the three island salt marshes, Cockle and Horse Islands, were located

within Core Sound and the third, Twin Island, was located within Back Sound, North Carolina, USA (Fig. 4-1). Site selection criteria included observed similarities in vegetation at equivalent tidal elevation, similar salinity and topographical slopes, and lack of dendritic rivulet development (Chapters 2 and 3). Island salt marsh sites were at least 400 m away from the nearest other salt marsh. This distance was near the maximum suggested summer dispersal range for F. heteroclitus (Lotrich 1975).

Physical and Environmental Parameters

Salt marsh vertical range was measured at each site, as were area size, perimeters of the salt marsh sites and distance from island salt marshes to the nearest other salt marsh (see Chapter 2). Shallow and low salt marsh sub-habitats were also delineated (see Chapter 2).

Salinity and water temperature were measured for each site during each collection. Both salinity (as measured with a temperature compensated refractometer, accuracy of 0.1 ppt), and temperature were measured during eel pot sets used to collect nekton.

Depopulation and Re-colonization

The affect of salt marsh site location immigration of Fundulus heteroclitus and Lagodon rhomboides was examined by the removal of populations of these target species from three large salt marsh island sites. During late fall 2004 (after spawning season for F. heteroclitus and prior to recruitment of L. rhomboides), baited eel pots were set at 15 m intervals along the entire perimeter of the salt marsh islands; along the deep salt marsh

edge that comprised that region of the island most exposed to storm energies and the low salt marsh (two meters inside the marsh fringe) that comprised the most protected area of the islands. Baited eel pots were also set at 15 m intervals along the low salt marsh/shallow salt marsh break at each island. A mainland fringing salt marsh site at Gunning Hammock was sampled concurrently with the island collections using three replicate baited eel pots located along the perimeter of the low salt marsh and three located along the low salt marsh/shallow salt marsh break. These collections were conducted at Gunning Hammock in order to monitor F. heteroclitus and L. rhomboides abundance during collection periods at the island salt marshes. Total eel pot replication per set was 41, 17 and 25 at Cockle, Horse and Twin Islands, respectively, and six at Gunning Hammock

Because the main objective of this sampling was to attract and collect all Fundulus heteroclitus and Lagodon rhombodies in the island salt marsh habitats and depopulate these sites of these species, eel pots were baited with 315 g of dry dog food. Baited eel pots have a higher attraction (Reebs et al. 1995) and retention (Whitelaw et al. 1991) compared to un-baited pots. Sufficient bait was used to ensure the bait supply was not exhausted over the 24 hour set duration. Eel pots were 80 cm in length, 22.5 cm in diameter, were constructed of 0.5 cm bar mesh and had conical capture ends that were positioned inward with 6 cm long by 3 cm wide capture openings (Halpin 1997; 2000; Kneib and Craig 2001). Eel pots were set during the morning, allowed to fish 24 hours and then retrieved.

All individuals collected at the islands were enumerated by species and standard length (SL) measured. These individuals were then removed from the site. Eel pots were

reset and collections continued at a site until no catch was observed for the target species during three consecutive collections at a site, or the population of target species had been reduced to significantly lower levels over three consecutive catches per unit effort (CPUE) as indicated by linear regression comparison to initial CPUE. For the reference site, individuals collected were enumerated by species and released live back at the point of collection so that potential reduction in catch at Gunning Hammock was not attributable to depopulation but reflected natural variability in abundances. During the initial collection at Gunning Hammock (depopulation of the islands), all, or if numerically abundant, a randomly selected subsample of at least 20 individuals for each species was measured (SL) from each eel pot.

Once sites were depopulated of target species, re-colonization was monitored through periodic eel pot sampling. The three island salt marshes were fished in the same manner and intensity as used during the depopulation work phase using eel pots approximately two months after depopulation, and then every other month through the end of peak spawning season for Fundulus heteroclitus the following year (July).

During the re-colonization phase, Fundulus heteroclitus and Lagodon rhomboides at the island and reference sites were enumerated and measured (SL). If numerically abundant, a random subsample of at least 20 individuals for each species from each eel pot were measured. Individuals collected were released live back at the point of collection until the final collection in July, during which all individuals collected were enumerated and measured.

In addition to the regular re-colonization sampling, test samplings at Horse Island were conducted 25 and 26 days after completion of the depopulation efforts for that site

while depopulation work was still ongoing at other sites. Four eel pots were placed along the low salt marsh/shallow salt marsh break and all individuals collected were enumerated, measured (SL), and released live back to the site of collection.

Statistical Analysis

Because of the preponderance of zero catches, comparisons between the initial CPUE at the islands to subsequent re-colonization CPUEs, and final observed CPUE during the depopulation phase to initial, high and subsequent re-colonization CPUEs relied on a conservative non-parametric method of analysis, the Wilcoxon Two Sample Test (Sokal and Rohlf 1981). Similar analyses were used for comparable times at the mainland site.

Linear Regression Analysis (regression analysis) was utilized in conjunction with the Wilcoxon Two Sample Test analyses to examine the effectiveness of depopulation for each island site. Regression analysis was used to determine if three consecutive samplings had regression line slopes that did not differ from zero, indicating no further significant reduction in the population might be expected. Data used in regression analysis were tested for normality using the Shapiro-Wilk test (Shapiro and Wilk 1965; Sen et al. 2003). If data were found not to be normal, data were ln (x + 1) transformed and again tested to assure data conformity.

Population size class frequency similarity among initial depopulation collections for all sites (highest for Twin Island due to few individuals initially encountered) and the final re-colonization collection were determined using the non-parametric Kolmogorov-Smirnov two sample test (Sokal and Rohlf 1981; Kneib and Craig 2001). In addition,

Wilcoxon Two Sample Tests were used to assess whether mean size of the treatment comparisons differed significantly from one another (Sokal and Rohlf 1981; Kneib and Craig 2001). These tests were also used for comparisons between island sites and the mainland for comparable time periods, and for observed immigrants to Horse Island compared to initial size class frequency distribution and the mainland site. For all test comparisons minimum significance level was $p = 0.05$.

## RESULTS

### Physical and Environmental Parameters

Gunning Hammock was substantially larger than island sites sampled for total salt marsh, high salt marsh, and low salt marsh habitats (Table 4-1). Cockle Island was the largest island site in terms of total, low and high salt marsh habitat area, and was the nearest to other salt marsh areas. Horse Island was most distant from other salt marshes and the smallest in terms of low salt marsh and total salt marsh, while Twin Island had the least amount of high salt marsh habitat (Table 4-1). Gunning Hammock had the lowest vertical relief in terms of observed salt marsh vegetation vertical range with that at Twin Island being the greatest (Table 4-1).

Temperature measurements at each site demonstrated predictable seasonal fluctuations based on collection time period (Fig. 4-2). Salinity measurements demonstrated periodic fluctuations, but overall trends were similar among the four collection sites (Fig. 4-2).

Depopulation Phase

Significant reductions in Fundulus heteroclitus CPUE between the initial and final depopulation collections were only observed for Horse Island (Table 4-2, Figs. 4-3a, b). Fundulus heteroclitus population size at Twin Island was initially observed to be depleted and no significant differences in CPUE compared to the last depopulation collection was observed (Table 4-2, Figs. 4-4a, b). While increases in catch, suggestive of colonization events, were evident during the depopulation collections at Twin Island, a comparison of abundance during peak CPUE and first and last CPUEs at Twin Island did not show significant differences in abundance between these collection events. At Cockle Island a pattern contrary to depopulation efforts was observed with significantly greater CPUE observed during the final depopulation collection compared to the initial collections (Table 4-2, Figs. 4-5a, b). Between the initial and final depopulation efforts, variable pulses of increased and decreased CPUE occurred, suggestive of pulsed colonization events (Table 4-2, Figs. 4-5a, b). While no significant difference in CPUE was apparent between the last collection and that of the highest catch peak, significantly higher CPUE was observed during the last compared to the initial collection (Table 4-2, Figs. 4-5a, b) at Cockle Island. No significant reduction in F. heteroclitus CPUE was evident at Gunning Hammock between the initial and final collections during the island depopulation phase (Fig. 4-5a). Between the initial and final island depopulation samplings, variable pulses of increased CPUE occurred suggesting pulsed F. heteroclitus recruitment (Fig. 4-5a).

Significant reductions in Lagodon rhomboides CPUE between the initial and final depopulation collections were observed for Horse Island (Table 4-2, Figs. 4-6a, b), and

between initial and highest catches compared to final CPUE at Cockle Island (Table 4-2; Figs. 4-8a, b). Lagodon rhomboides CPUE at Twin Island was initially observed to be low and no significant differences in catches compared to the last depopulation collection was observable (Table 4-2, Figs. 4-7a, b). Significant reductions in L. rhomboides catch between the initial and final depopulation collections was also observed for Gunning Hammock (Fig. 4-8a).

Linear regression analysis indicated that the slope of the last three collection points for all three island sites did not significantly differ from zero for Fundulus heteroclitus (Figs. 4-3b, 4-4b and 4-5b) and Lagodon rhomboides CPUE (Figs. 4-6b, 4-7b, and 4-8b). However, linear projection of the predicted slope lines revealed that while flat line conditions were apparent for L. rhomboides CPUE at Horse, Twin and Cockle Islands (Figs. 4-6b, 4-7b and 4-8b), and for F. heteroclitus CPUE at Horse and Twin Islands (Figs. 4-3b and 4-4b), a negative slope was obvious for F. heteroclitus CPUE at Cockle Island (Fig. 4-5b). While the slope of the line for F. heteroclitus CPUE at Cockle Island did not significantly differ from a zero slope (possibly due to low replication), the presence of this negative slope line combined with an observed significant increase in the final depopulation F. heteroclitus CPUE compared to initial CPUE at Cockle Island (Table 4-2) suggested that reduction in F. heteroclitus CPUE was not successful at this site (Fig. 4-5b).

Re-colonization Phase

Significant increases in Fundulus heteroclitus CPUE were evident at Horse Island between the final depopulation collection in November 2004 and re-colonization catches

during March, May and July 2005 (Table 4-2, Figs. 4-3a, b). No significant differences were evident between the initial depopulation F. heteroclitus CPUE and re-colonization catches at Horse Island during March and July 2005 (Table 4-2). Significant increases in F. heteroclitus CPUEs were evident at Twin Island between the final depopulation collection in December 2004 and re-colonization collections during January, March and July 2005 (Table 4-2, Figs. 4-4a, b). Significant increases in F. heteroclitus CPUE also were evident at Twin Island between the initial depopulation CPUE in November 2004 and re-colonization catches during January, March and July 2005 (Table 4-2). For Cockle Island no significant increase in population size for F. heteroclitus associated with re-colonization was evident among the final depopulation CPUE and re-colonization catches from January, March, May and July 2005 (Table 4-2, Figs. 4-5a, b). However, significant increases in F. heteroclitus CPUE were evident at Cockle Island during re-colonization for May and July 2005 compared to the initial depopulation catch in October 2004 (Table 4-2). For Gunning Hammock, significant differences in F. heteroclitus CPUE between December 2004 (the time of the final island depopulation collections) and re-colonization period catches were only observed for July 2005 (Fig. 4-5a). Significant increases in F. heteroclitus during the re-colonization phase compared to the initial October 2004 sampling were evident at Gunning Hammock during July 2005 (Fig. 4-5a).

Significant increases in Lagodon rhomboides CPUE were observed during the re-colonization phase compared to final depopulation collections at all island sites and the Gunning Hammock reference site (Table 4-2, Figs. 4-6a, b, 4-7a, b and 4-8a, b). Significant increases in L. rhomboides CPUE were evident at Horse and Cockle Islands between the final depopulation collection in November and December 2004, respectively,

and the re-colonization collection in July 2005 (Table 4-2, Figs. 4-6a, b and 4-8a, b). Similarly, significant increases in L. rhomboides CPUE were evident at Twin Island and Gunning Hammock between the final depopulation collection in December 2004 and May and July 2005 re-colonization collections (Table 4-2, Figs. 4-7a, b and 4-8a).

Significantly lower Lagodon rhomboides CPUEs were observed for re-colonization phase collections during January, March and May 2005 at Cockle Island (Table 4-2), and January and March 2005 at Gunning Hammock (Fig. 4-8a) compared to initial depopulation collections during October 2004. Similarly, significantly lower L. rhomboides catches were observed for re-colonization phase collections during January, March and May 2005 at Horse Island compared to initial depopulation collections during November 2004 (Table 4-2). A contrary pattern was observed for Twin Island with no significant differences apparent between the initial depopulation collections from November 2004 compared to re-colonization phase collections during January and March 2005, likely due to low initial CPUE observed during the depopulation phase (Table 4-2). A significant increase in L. rhomboides CPUE was evident at Twin Island during the May and July 2005 re-colonization collections compared to the initial November 2004 depopulation collection CPUE (Table 4-2).

Estimate of Potential Immigrants to Horse Island

Fundulus heteroclitus immigration to Horse Island during the 25 day break in depopulation effort was estimated using suspected F. heteroclitus immigration increases observed at nearby Cockle Island during the same time period (Table 4-2). This estimate was performed to ascertain if the colonization rate of immigrants measured at Horse

Island through dispersal were reasonable based on that estimated for a nearby island in a similar estuarine setting. For this calculation estimate, the observed total at Cockle Island during November 11th of 78 individuals collected was considered as an origin for continued reduction following the pattern observed at Horse Island for a similar catch reduction observed during November 4th of 55 individuals. The sequential depletion for Cockle Island was then considered to be 78, 47, 8, 3, 3, 3, 3, 3, 3, and 3 for total F. heteroclitus collected to represent a remaining remnant population totaling 91, with 3025 individuals (0.29 individuals $m^{-2}$) considered to be immigrants to Cockle Island during this 25 day time period. Based on this immigration estimate a total of 833 (0.29 x 2,873) individuals might have immigrated to Horse Island during the 25 day collection break. This estimate is consistent with 421 F. heteroclitus immigrants collected in limited sampling (Table 4-2) and the estimated 565 total individuals that could be expected to have occurred at Horse Island during this limited re-sampling based on a sequential fish down of 223, 198, 76, 46, 14, 6, and 2 individuals.

Size Frequency Distributions

Comparison of Fundulus heteroclitus population size frequency distribution for all three islands and the reference mainland revealed that fall populations differed significantly ($p < 0.0001$) from the summer populations with larger size classes contributing more to summer populations than during the fall (Figs. 4-9a-d). However, differences between fall and summer population mean size were not as extreme for individuals measured at Cockle Island (50.2 mm, fall and 50.8 mm, summer) and Gunning Hammock (46.1 mm, fall and 47.5 mm, summer) compared to Horse (46.6 mm,

fall and 56.2 mm, summer) and Twin (49.2 mm, fall and 60.2 mm, summer) Islands (Figs. 4-9a-d). For Lagodon rhomboides an opposite trend was observed at the island sites with significantly ($p < 0.0001$) smaller individuals and mean size predominating during the summer (55.4 mm at Cockle, 54.1 mm at Horse and 41.0 mm at Twin Islands) and larger individuals during the fall (68.8 mm at Cockle, 68.9 mm at Horse and 56.9 mm at Twin Islands) (Figs. 4-10a-c). However, no significant ($p > 0.05$) difference in L. rhomboides population size distribution or mean size between fall (46.5 mm) and summer (49.0 mm) was observed at the mainland reference site, Gunning Hammock (Fig. 4-10d).

Fundulus heteroclitus size class frequency distribution and mean size comparisons between individual island sites and Gunning Hammock for initial collections (high in the case of Twin Island) and during the final re-colonization collection revealed that island sites differed significantly from Gunning Hammock during both fall and summer (Figs. 4-11a-d, 4-12a-d and 4-13a-d) with larger mean size observed at the islands compared to Gunning Hammock during comparable time periods. Similarly, Lagodon rhomboides size class frequency distribution and mean size comparisons between individual island sites for initial depopulation collections (highest in the case of Twin Island) and Gunning Hammock, and during the final re-colonization collection, showed that island site populations differed significantly from the population at Gunning Hammock during fall and summer (Figs. 4-14a, b, 4-15a, b, and 4-16a, b). Larger mean size was observed at Cockle and Horse Islands during both fall and summer, and at Twin during the fall compared to Gunning Hammock, and smaller individuals at Twin Island compared to Gunning Hammock during the summer.

Size distribution comparisons of Fundulus heteroclitus at Horse Island during the fall depopulation collections, including examination of size class distribution of immigrants collected from the site 25 days after initial depopulation, suggested immigrant commonality with the population sampled at Gunning Hammock (Figs. 4-17a-c). Both population size class frequency distribution and mean size from the initial depopulation collections at Horse Island (46.6 mm) differed significantly from those observed at Gunning Hammock (46.1 mm) ($p < 0.0001$, and $p = 0.0042$ respectively) (Fig. 4-17a). Comparison of initial population size class frequency distribution and mean size at Horse Island compared to immigrants to Horse Island also showed the population size class frequency structure and mean size (initial = 46.6 and immigrant = 44.3 mm) significantly differed ($p < 0.0001$) for F. heteroclitus (Fig. 4-17b). Similarity comparisons for Horse Island F. heteroclitus immigrants to the F. heteroclitus population at Gunning Hammock revealed no significant differences between the two populations for size class frequency distribution nor mean size (Fig. 4-17c).

## DISCUSSION

Depopulation efforts produced low abundances of Fundulus heteroclitus at two island salt marsh sites. Decreased abundances of Lagodon rhomboides occurred over time at all three island sites and the mainland reference, possibly due to decreasing water temperatures causing L. rhomboides to move to offshore areas (Muncy 1984). During the depopulation phase it was evident that significant reductions in both F. heteroclitus and L. rhomboides populations were produced at Horse Island, while F. heteroclitus remained abundant at the mainland site. Similar efforts to reduce fish population size for

re-colonization estimates have also been attempted for hard bottom reefs (Parker and Greene 1999) and tide pools (Polivka and Chotkowski 1998; Griffiths et al. 2004) with varying degrees of success. While depopulation efforts were fully successful at Horse Island for both F. heteroclitus and L. rhomboides, populations for both species were already low at Twin Island and no further catch reductions were detectable. The F. heteroclitus population at Cockle Island could not be fished down, probably due to both island size and apparent immigration of F. heteroclitus during the depopulation phase, though significant reductions in L. rhomboides were observed. Similar to these observations, Parker and Greene (1999) noted rapid immigration in predator fish species at hard bottom reefs during depopulation efforts that complicated removal efforts during their study for re-colonization estimations.

Immigration of Fundulus heteroclitus to the islands happened within weeks to months of depopulation. Similar rapid re-colonization of fishes to depopulated hard bottom reefs (Parker and Greene 1999) and tide pools (Polivka and Chotkowski 1998; Griffiths et al. 2004), as well as rapid colonization of reef fishes to artificial reefs set among existing coral reefs (Stone et al. 1979) have also noted. During the depopulation phase a test sampling at Horse Island, using less than one-forth of the collection locations used for the depopulation phase, was conducted within one month of the final depopulation collections. This sampling revealed that significant numbers of F. heteroclitus were present at the island, and that over one-half of the total number of F. heteroclitus individuals removed from the island during the depopulation phase had already repopulated Horse Island. Calculated immigrant totals for Horse Island based on these limited test samplings were also comparable to calculated potential immigrant

totals used to determine if the colonization rate of immigrants measured at Horse Island through dispersal were reasonable during the 25 day interval between collections based on suspected immigration of F. heteroclitus, on a unit area basis, to nearby Cockle Island. Although this comparison assumed equivalent colonization potential for both islands, such was not unreasonable given the similarity of the shallow water flats that surround both islands, connecting them to the mainland, and their similar geographic position within the estuary. The rapid re-establishment of a significant F. heteroclitus population at Horse Island indicated that dispersal/immigration was occurring during the depopulation time period. Collections at other sites during the same time period also suggested that immigration was occurring; e.g., an evident population increase at Twin Island and periodic pulsed increases in CPUE for F. heteroclitus at Cockle Island.

Additional evidence for dispersal to island salt marsh habitat by Fundulus heteroclitus was provided by the similarity of the population size class structure of the immigrants to the mainland population for the same time period. Further, the similarity between the immigrant F. heteroclitus populations observed at Horse Island and Gunning Hammock during the fall suggested mainland population origin for the immigrants at Horse Island as predicted by the mainland-island metapopulation model (Harrison and Taylor 1997). Similar population dispersal between mainland and islands due to immigration was also probably occurring at the other salt marsh island sites as well.

The rapid re-establishment of the population at Horse Island during the fall indicated that F. heteroclitus is capable of immigrating to isolated island salt marshes across substantial open water distances of over 800 m and that movement of large numbers can occur in a relatively rapid time period. For Lagodon rhomboides,

immigration to the island and mainland was not observed until the population had reached size classes large enough to be targeted by the collection gear, during May and July (Chapter 2), but it probably had colonized these habitats by late winter (Warlen and Burke 1990; Meyer et al. 1996; Chapter 3).

The consistent difference in Lagodon rhomboides size class structure between fall and summer collections for the island salt marshes suggested the young of year (YOY) use of these sites was considerable during summer. Predictable cohort size increase due to growth during the summer was the likely cause of the significantly larger L. rhomboides in the fall collection of the previous year (Muncy 1984). The lack of significant difference for L. rhomboides fall and summer size class distribution comparison at the mainland site suggests size class specific use of this habitat type. This might be related to the inability of small individuals to traverse the expansive shallow water flat adjacent to this site (Knieb and Wagner, 1994), and limitation of larger individuals to deeper water areas (Chapter 2).

An opposite seasonal pattern was observed for Fundulus heteroclitus at the island salt marshes, which on average demonstrated consistently larger sized individuals at all sites during summer compared to the fall. The more similar population structure profile for F. heteroclitus between the island and mainland during the fall time period (dominated by YOY size classes) compared to summer (dominated by year 1 and year 1+ size classes) (Chapter 3), may indicate that isolated islands of the size examined during this study might be capable of producing limited YOY that eventually contribute to the overall F. heteroclitus populations, and thus might be able to support limited F. heteroclitus populations. Alternatively, it is also possible that seasonally pulsed

population dispersal within the estuary might have occurred prior to depopulation efforts and that the population size structure at the island sites are a conglomerate of any existing remnant island populations and immigrants from mainland salt marshes, such as Gunning Hammock. The possibility of F. heteroclitus distributional changes correlated with time periods of reduced predator concentrations has been suggested by Halpin (1997; 2000) and supported by observations from other studies (Chapters 2 and 3). A combination of larger size classes of F. heteroclitus persisting on the salt marsh islands compared to mainland salt marshes (Chapter 2), and size dependant colonization of more distant habitats by larger individuals capable of reaching them (Parker 1971), would create a shift toward larger overall size classes at the islands. Less difference in size structure between the islands and mainland would be apparent during the fall when dispersal increases with a predicted shift towards larger overall size classes on islands occurring during late spring/early summer when movement between island and mainland salt marsh might become restricted (Chapters 2 and 3). Knieb and Wagner (1994) noted more extensive movement of larger F. heteroclitus than smaller YOY within salt marsh habitat and suggested that larger individuals were better able to range throughout the salt marsh and utilize a wider range of habitat during one tidal cycle.

Seasonal dynamics in the dispersal and colonization behavior of Fundulus heteroclitus was suggested by overall fall summer size class frequency distribution. While the size class distribution for Horse Island and Twin Island differed significantly from the mainland (Gunning Hammock) during the fall depopulation phase, disparities between the mainland and islands were more striking during the summer. The consistency in the magnitude of divergence in size class structure for F. heteroclitus

between fall and summer for both depopulated islands compared to the mainland, with increased disparities in F. heteroclitus size class structure between the mainland and island salt marshes during the summer, suggests that immigration to these habitats was significantly reduced during the summer. Reduced juvenile F. heteroclitus production at the island compared to mainland salt marshes (Chapter 3), increased predation pressure for islands (Chapters 2 and 3), and reduced food resources (Raichel et al. 2003) also likely contributed to the observed size class patterns.

Because significant increase in the Fundulus heteroclitus abundance at Gunning Hammock was only observed between the last fall collection and seven months later during the last re-colonization collection, when substantial YOY contribution was evident, YOY driven population maintenance is surmised. However, adult immigration was essentially maintaining island F. heteroclitus populations rather than YOY recruitment based on significant increases prior to the recruitment time period. Similar population maintenance patterns and processes have been observed for the crested newt (Triturus cristatus) populations among fresh water ponds (Miaud et al.1993) and the re-establishment of fishes to depopulated tide pools (Polivka and Chotkowski 1998; Griffiths et al. 2004). The disparity in evident YOY contribution to island and mainland populations (also see Chapters 2 and 3) further indicates that salt marsh islands are primarily being sustained by immigrants that colonize the island salt marsh sites during fall and winter time periods, with these island salt marshes acting as sinks for estuarine F. heteroclitus populations.

Periodic colonization by Fundulus heteroclitus occurred at Twin Island during the depopulation effort, after initial collections had indicated depleted F. heteroclitus

populations. These colonization episodes are suspected to have been triggered by cold weather pulses that initiated mass dispersal of F. heteroclitus into new habitat areas just as cold fall weather is known to initiate mass movement of L. rhomboides from coastal estuaries to oceanic waters (Muncy 1984). The periodic recruitment of F. heteroclitus to Twin Island eventually produced population increases at the island. However, long term support for the F. heteroclitus population at Twin Island was questionable as the YOY that did immigrate during the fall declined in number through winter, spring and summer. This overall decline in F. heteroclitus abundance at Twin Island might be due to insufficient quantity and quality of support habitat (Chapters 2 and 3), and seasonal isolation from other salt marsh habitats due to increased predator presence (Chapter 3) restricting F. heteroclitus movement to and from the island (Halpin 2000) via shallow water movement corridors (Ruiz et al. 1993; Chapters 2 and 3).

Halpin (2000) noted differential salt marsh habitat use by Fundulus heteroclitus based on time of year, with more widespread usage of shallow, more exposed mudflat habitat during time periods when predation might be low. Such differential use patterns might explain re-colonization of salt marsh islands by F. heteroclitus population during colder times of the year (Chapter 2). Increased movement is suspected to occur during the late fall and early winter (Chapter 2), associated with cold fronts producing sharp decreases in water temperature. The sharp decreases in water temperature are suspected to initiate the mass movements of F. heteroclitus to disperse among other habitats utilizing shallow water flats, banks and bars, colonizing these new habitats during periods of low predator abundance (Chapter 3).

Pulsed colonization during the late fall through late winter by salt marsh-

dependant resident nekton, such as Fundulus heteroclitus and Cyprinodon variegatus, would help to explain the sudden increase in abundances of these species within the large and small island salt marshes during late winter/early spring fyke net collections (Chapter 3). It would also partially explain subsequent reductions in abundance over the course of summer as movement corridors became unusable, due to predator increases (Chapter 3), and isolated populations became depleted (Chapter 2) due to local resource issues including food availability (Raichel et al. 2003), predator refuge (Chapter 3) and habitat related recruitment success of YOY (Chapter 3).

There are two crucial time periods for Fundulus heteroclitus population maintenance, one during the late spring and early summer spawn and recruitment period (Talbot and Able 1984; Abrams 1985) and the second during the dispersal period for adult-sized F. heteroclitus during the late fall through late winter (Chapter 3). The emigration/immigration phase of this distribution pattern for F. heteroclitus might be more significant for maintaining the geographic distribution for this species among salt marsh islands, while the breeding/recruitment phase is critical for local population maintenance in large contiguous salt marshes. By contrast, for many transient species, such as Lagodon rhomboides, the crucial processes of recruitment and dispersal occur simultaneously during the late fall to late winter (Warlen and Burke 1990). During these times low predator levels (Chapter 3) combine with recruitment to enhance both geographical and local population maintenance on island salt marshes. It was evident that both F. heteroclitus and L. rhomboides are capable of seasonally colonizing salt marsh habitat at the level of isolation investigated during this study. While this was expected for L. rhomboides based on known habitat use patterns and the transient nature

of this pelagic spawner, this was unexpected for F. heteroclitus. The ability of F. heteroclitus to disperse and quickly colonize new salt marsh habitats over 800 m of open water, as observed here, is twice that originally suggested by Lotrich (1975) and Teo and Able (2003). This indicates that its dispersal ability has been underrated and that colonization of island habitats is probable for seed populations. However, the resilience of the isolated salt marsh immigrant populations is subject to habitat quantity and quality and periodic extinction events might occur for populations that become isolated during times of the year when movement between habitats is restricted due to predation related mortality. Thus, while habitat quantity and quality issues remain in terms of determining population success, it was evident that at the scale measured here, salt marsh degree of isolation did not significantly influence the ability of L. rhomboides and F. heteroclitus to recruit to and colonize salt marsh habitat.

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Table 4-1. Landscape attributes for island and mainland salt marsh sites sampled.

| Marsh Site | Vertical Relief (cm) | Total Marsh Size ($m^2$) | Low Marsh Size ($m^2$) | Shallow Marsh Size ($m^2$) | Distance to Nearest Marsh ($m^2$) |
|---|---|---|---|---|---|
| Horse Island | 77.6 | 2,873 | 1,719 | 1,154 | 844 |
| Twin Island | 94.8 | 3,479 | 3,045 | 434 | 840 |
| Cockle Island | 90.6 | 10,463 | 3,635 | 6,828 | 431 |
| Gunning Hammock | 66.6 | 76,100 | 19,325 | 56,775 | 0 |

Table 4-2. Collection totals/mean per trap for each collection date and site for Fundulus heteroclitus (FUH) and Lagodon rhomboides (LAR). Catch in parentheses for Horse Island during 12/06 and 12/07 were test samplings using only four eel pots to assess potential re-colonization. — indicates that no collections were made at that site during that collection date. Asterisks designate that collection means per trap for initial, high or specific re-colonization collections are significantly different from the final depopulation mean for that site and species ($p \leq 0.05$). An (N) designates that collection means per trap for initial do not significantly differ from specific re-colonization collections for that site and species ($p > 0.05$)

| Collection Date (Set) | Cockle Island FUH | Cockle Island LAR | Horse Island FUH | Horse Island LAR | Twin Island FUH | Twin Island LAR |
|---|---|---|---|---|---|---|
| Depopulation Phase | | | | | | |
| 10/26 (1) | 149 / 3.63*N | 1004 / 24.49*N | — / — | — / — | — / — | — / — |
| 10/27 (2) | 257 / 6.27 | 1067 / 26.02* | — / — | — / — | — / — | — / — |
| 10/28 (3) | 189 / 4.56 | 1004 / 24.39 | — / — | — / — | — / — | — / — |
| 10/29 (4) | 301 / 7.34 | 784 / 19.98 | — / — | — / — | — / — | — / — |
| 10/30 (5) | 658 / 16.05 | 1003 / 24.32 | — / — | — / — | — / — | — / — |
| 11/02 (6) | 808 / 20.61 | 967 / 23.58 | 561 / 33.00*N | 464 / 27.29*N | — / — | — / — |
| 11/03 (7) | 625 / 15.12 | 574 / 14.00 | 143 / 8.41 | 390 / 22.94 | — / — | — / — |
| 11/04 (8) | 954 / 23.27 | 311 / 7.58 | 55 / 3.23 | 312 / 18.35 | — / — | — / — |
| 11/05 (9) | 352 / 8.58 | 112 / 2.73 | 33 / 1.94 | 138 / 8.12 | — / — | — / — |
| 11/08 (10) | 1170 / 28.54 | 7 / 0.17 | 11 / 0.65 | 49 / 2.88 | — / — | — / — |
| 11/09 (11) | 451 / 11.00 | 2 / 0.05 | 5 / 0.29 | 9 / 0.53 | — / — | — / — |
| 11/10 (12) | 602 / 14.68 | 3 / 0.07 | 2 / 0.12 | 10 / 0.59 | — / — | — / — |
| 11/11 (13) | 78 / 1.90 | 1 / 0.02 | 3 / 0.18* | 9 / 0.53* | — / — | — / — |
| 11/15 (14) | 159 / 4.17 | 2 / 0.05 | — / — | — / — | 1 / 0.04N | 6 /0.24N |
| 11/16 (15) | 326 / 7.95 | 1 / 0.02 | — / — | — / — | 3 / 0.12 | 7 /0.28 |
| 11/17 (16) | 317 / 7.73 | 0 / 0.00 | — / — | — / — | 2 / 0.08 | 3 /0.12 |
| 11/18 (17) | 587 / 14.32 | 0 / 0.00 | — / — | — / — | 11 / 0.44 | 1 /0.04 |
| 11/22 (18) | 207 / 5.05 | 0 / 0.00 | — / — | — / — | 3 / 0.12 | 13 /0.52 |
| 11/23 (19) | 182 / 4.44 | 1 / 0.02 | — / — | — / — | 16 / 0.64 | 12 /0.48 |
| 11/29 (20) | 203 / 4.95 | 0 / 0.00 | — / — | — / — | 4 / 0.16 | 7 /0.28 |
| 11/30 (21) | 309 / 7.54 | 0 / 0.00 | — / — | — / — | 4 / 0.16 | 9 /0.36 |
| 12/01 (22) | 322 / 7.85 | 2 / 0.00 | — / — | — / — | 2 / 0.08 | 7 /0.28 |
| 12/02 (23) | 504 / 12.29 | 0 / 0.00 | — / — | — / — | 3 / 0.12* | 3 /0.28* |
| 12/06 (24) | 230 / 5.60 | 0 / 0.00 | (223)/(55.75) | (0)/(0.00) | — / — | — / — |
| 12/07 (25) | 344 / 8.39* | 0 / 0.00* | (198)/(49.50) | (0)/(0.00) | — / — | — / — |
| Re-colonization Phase | | | | | | |
| 1/05 (26) | 141 / 3.44N | 1 / 0.02 | 69 / 4.06 | 0 / 0.06 | 28 / 1.12* | 0 / 0.00N |
| 3/05 (27) | 237 / 5.78N | 2 / 0.05 | 97 / 5.71*N | 1 / 0.00 | 40 / 1.60* | 3 / 0.12N |
| 5/05 (28) | 495 / 12.07 | 23 / 0.56 | 63 / 3.70* | 0 / 0.00 | 8 / 0.32N | 42 / 5.68* |
| 7/05 (29) | 644 / 15.71 | 578 / 13.85*N | 223 / 13.12*N | 442 / 26.00*N | 27 / 1.08* | 1581/63.28* |

Fig. 4-1. Site locations within Back and Core Sounds of North Carolina. Locations corresponding to the circle is the mainland salt marsh site, squares are island salt marsh sites.

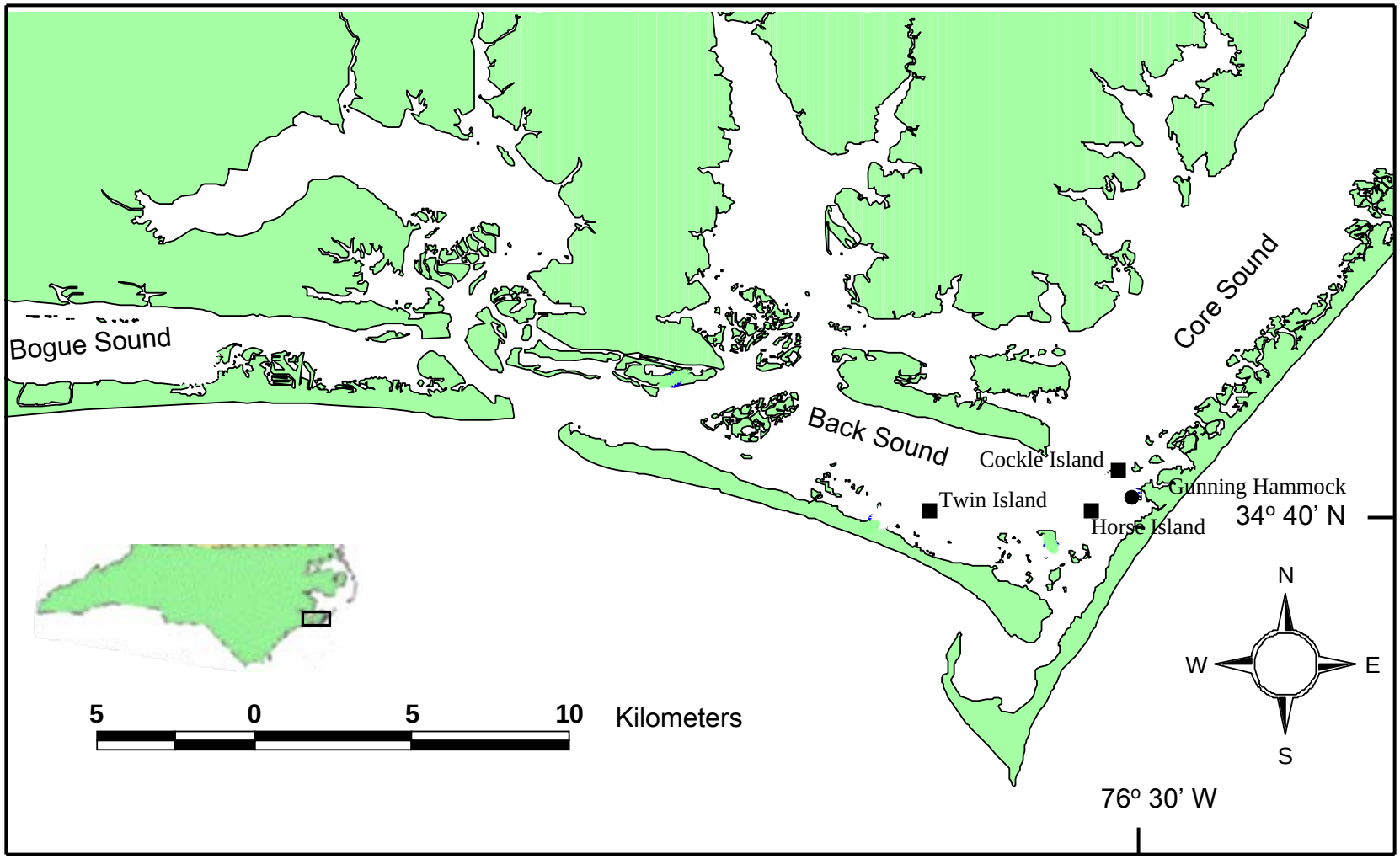
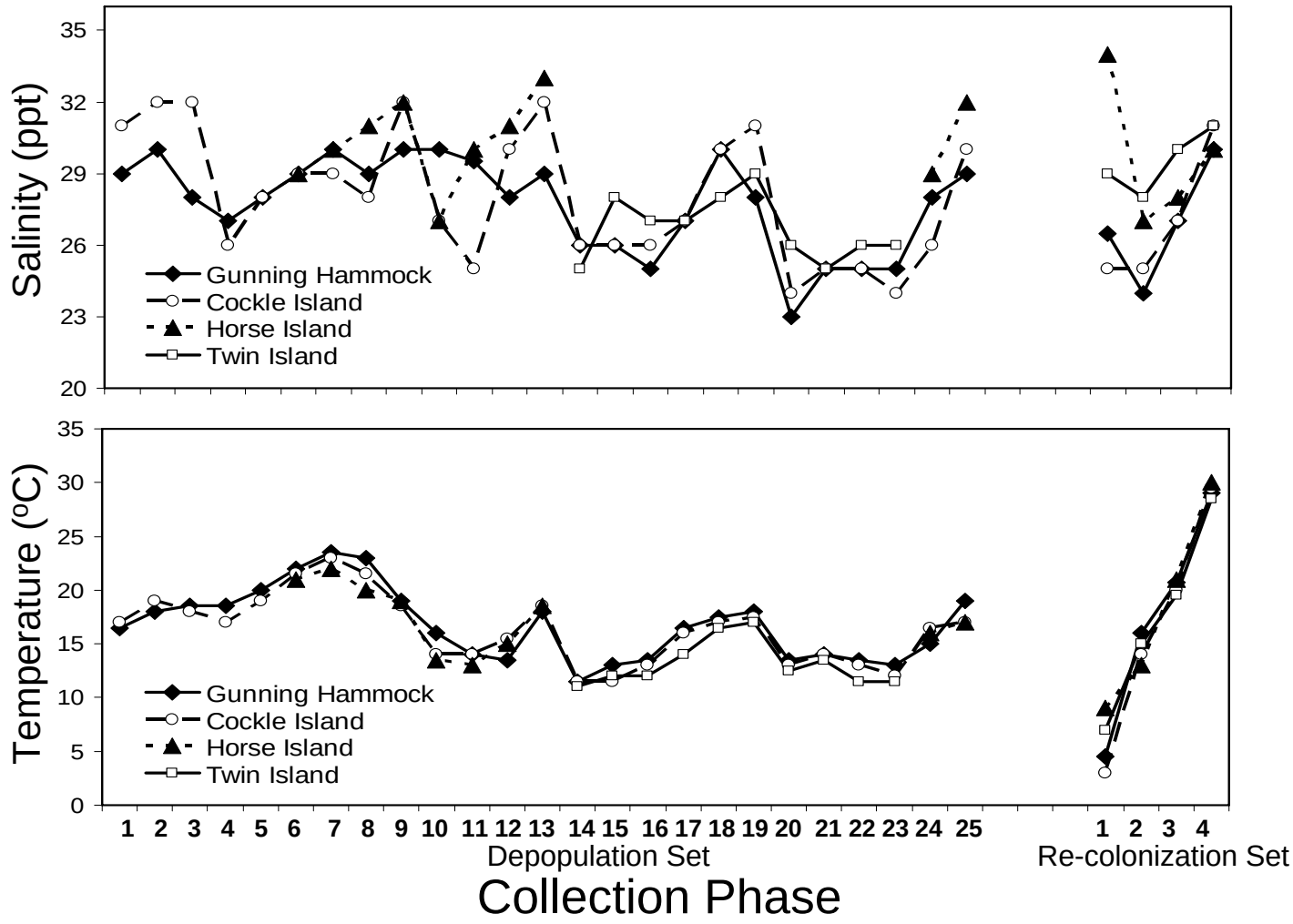
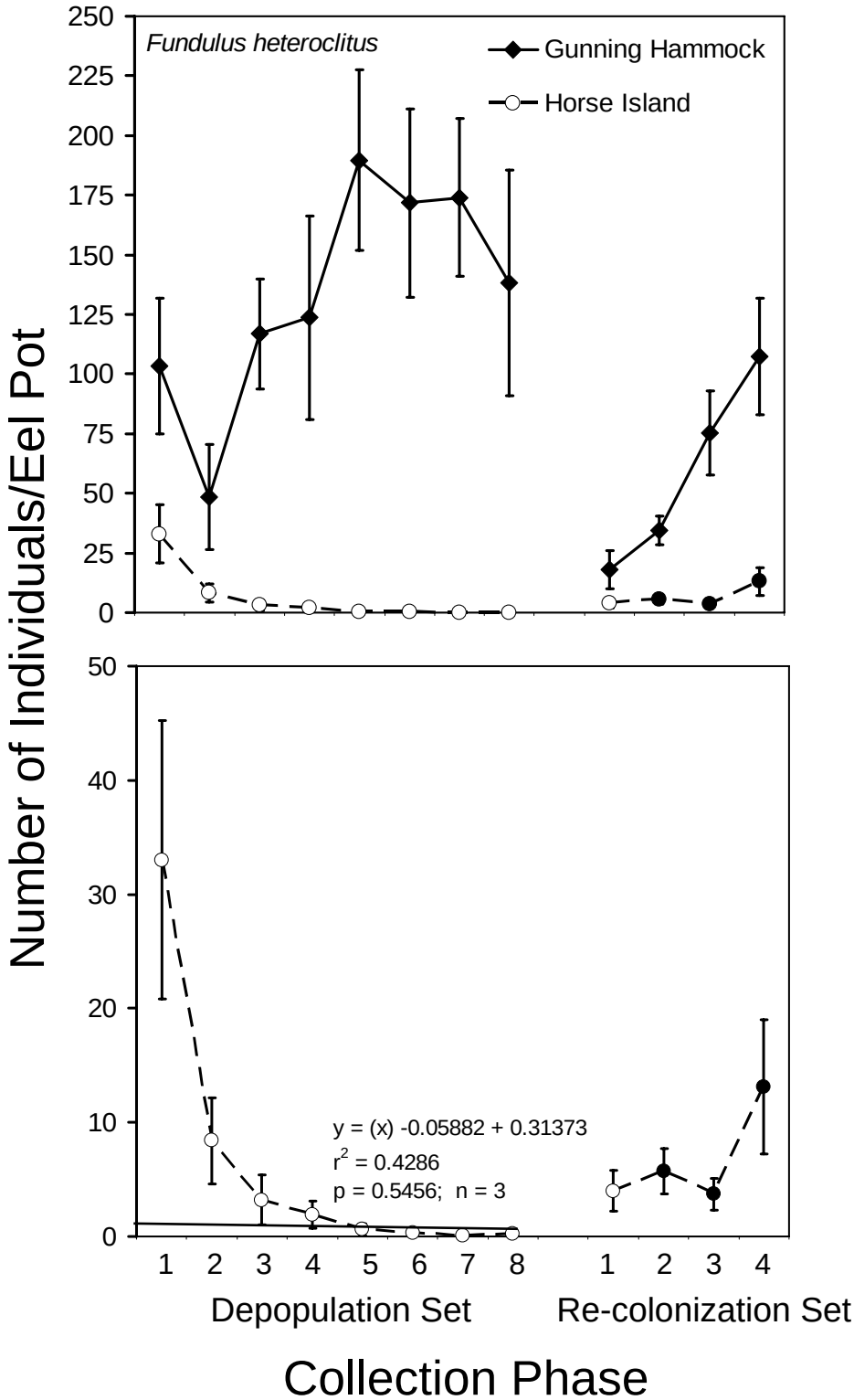

Bogue Sound
Back Sound
Core Sound
Cockle Island
Gunning Hammock
Twin Island
Horse Island
34° 40' N
76° 30' W
N
W
E
S
5 0 5 10 Kilometers

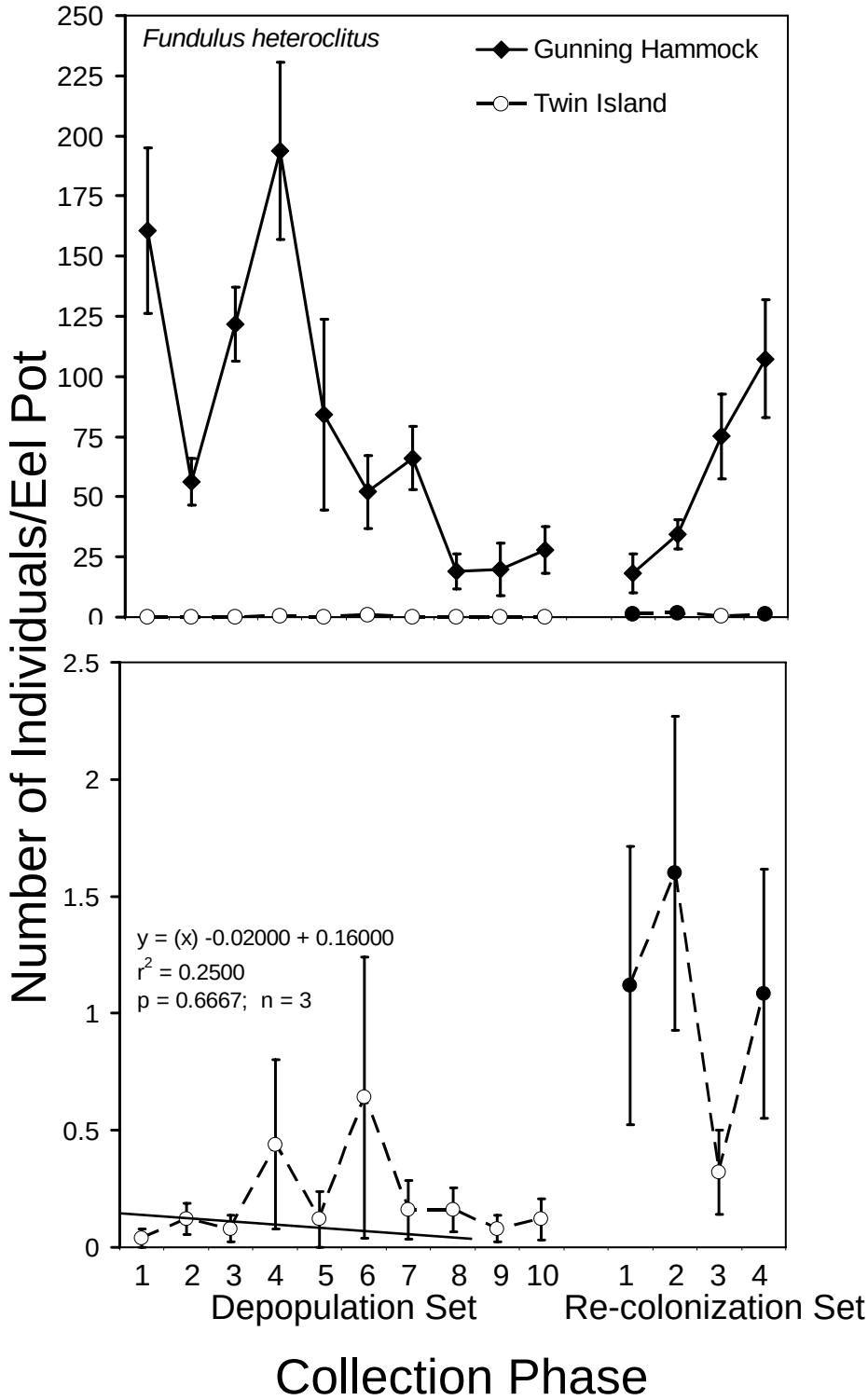
Fig. 4-2. Salinity (ppt) and water temperature (°C) during each collection set per site.


Salinity (ppt)
35
32
29
26
23
20
Gunning Hammock
Cockle Island
Horse Island
Twin Island
Temperature (°C)
35
30
25
20
15
10
5
0
Gunning Hammock
Cockle Island
Horse Island
Twin Island
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25
Depopulation Set
1 2 3 4
Re-colonization Set
Collection Phase

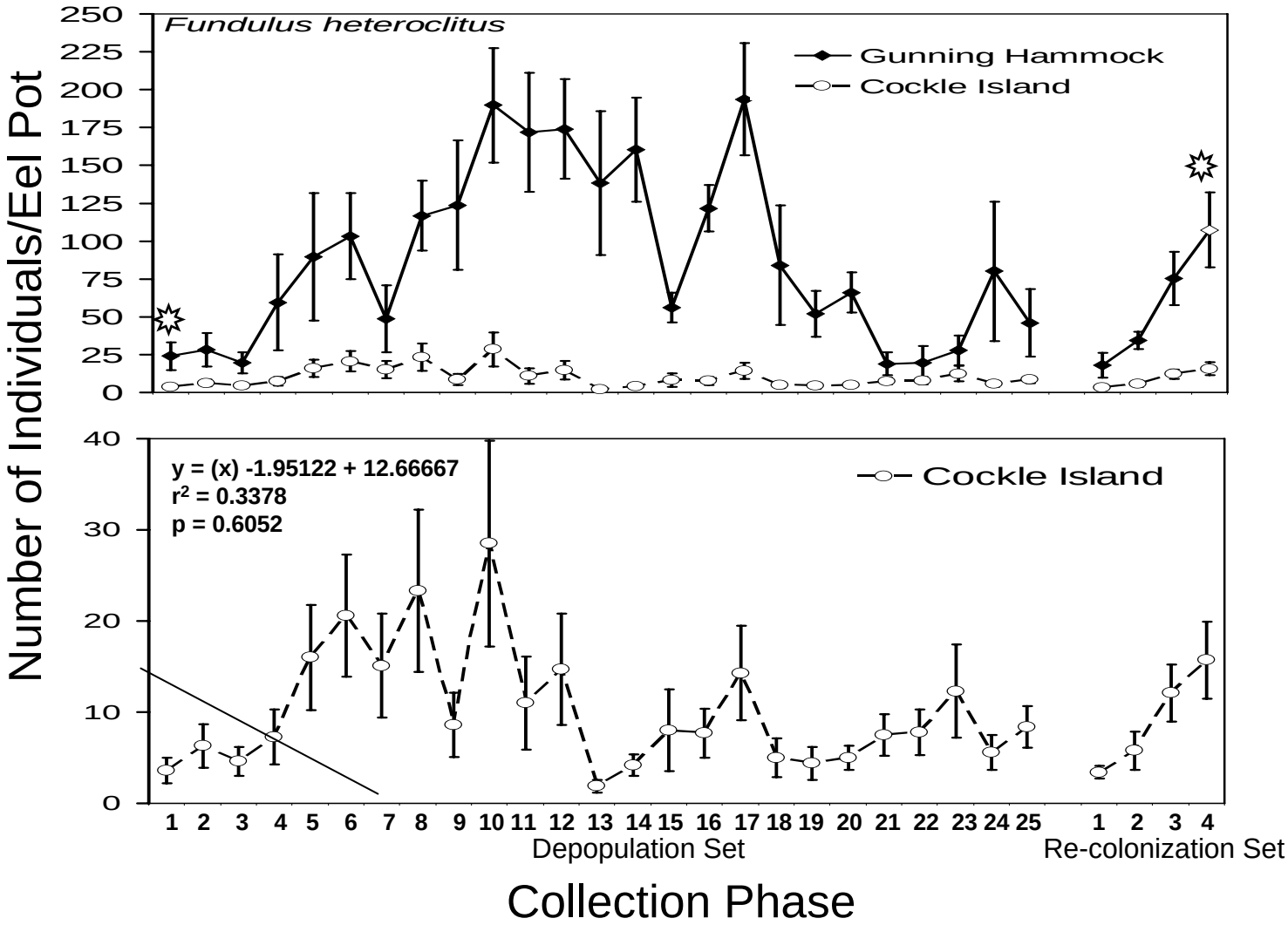
Figs. 4-3a,b. Mean number of Fundulus heteroclitus collected per eel pot (CPUE), per set for Gunning Hammock (a), and Horse Island (a and b) during the depopulation and re-colonization phases. For Horse Island, collection CPUEs during the re-colonization phase that are significantly different from the final depopulation phase collection CPUE are indicated by solid circles. Linear regression analysis formula and slope line are indicated for the final three depopulation phase sets at Horse Island. One standard error for CPUE is indicated for all sets, ($p \leq 0.05$).


Fundulus heteroclitus
Gunning Hammock
Horse Island
Number of Individuals/Eel Pot
250
225
200
175
150
125
100
75
50
25
0
50
40
30
20
10
0
y = (x) -0.05882 + 0.31373
r2 = 0.4286
p = 0.5456; n = 3
1 2 3 4 5 6 7 8
1 2 3 4
Depopulation Set
Re-colonization Set
Collection Phase

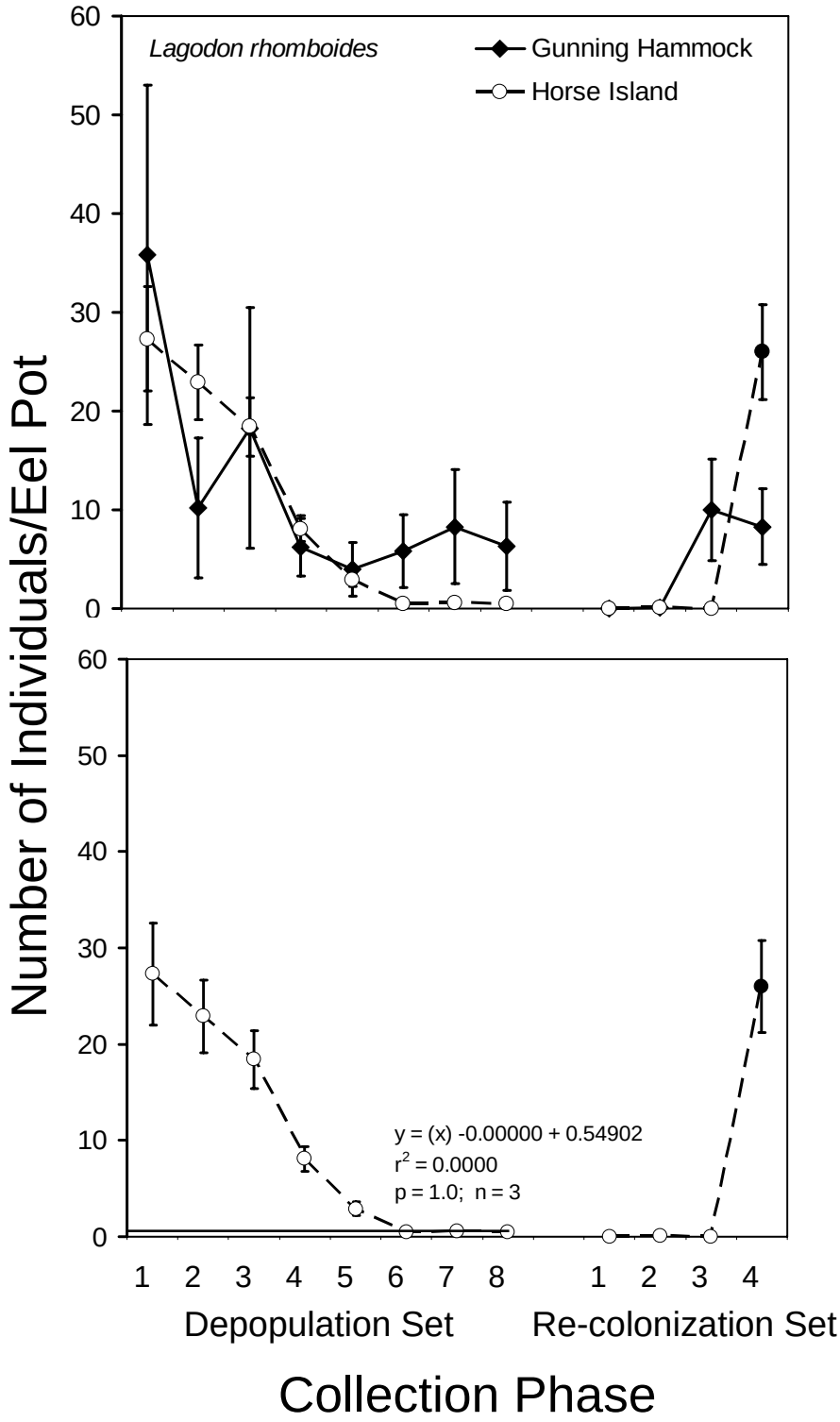
Figs. 4-4a,b. Mean number of Fundulus heteroclitus collected per eel pot (CPUE), per set for Gunning Hammock (a), and Twin Island (a and b) during the depopulation and re-colonization phases. For Twin Island, collection CPUEs during the re-colonization phase that are significantly different from the final depopulation collection phase CPUE are indicated by solid circles. Linear regression analysis formula and slope line are indicated for the final three depopulation phase sets at Twin Island. One standard error for CPUE is indicated for all sets, ($p \leq 0.05$).


Fundulus heteroclitus
Gunning Hammock
Twin Island
Number of Individuals/Eel Pot
250
225
200
175
150
125
100
75
50
25
0
2.5
2
1.5
1
0.5
0
y = (x) -0.02000 + 0.16000
r2 = 0.2500
p = 0.6667; n = 3
1 2 3 4 5 6 7 8 9 10
1 2 3 4
Depopulation Set
Re-colonization Set
Collection Phase

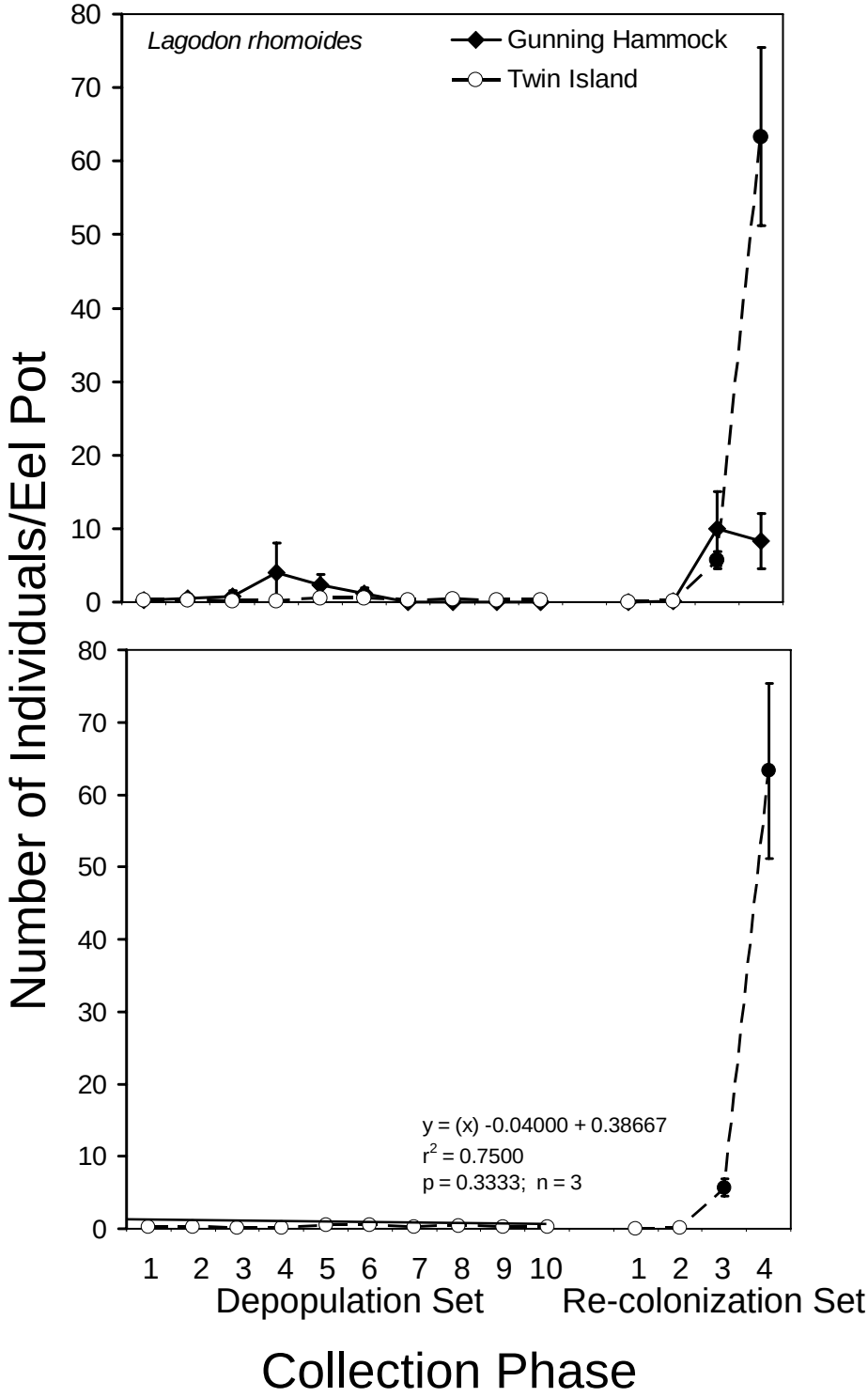
Figs. 4-5a,b. Mean number of Fundulus heteroclitus collected per eel pot (CPUE), per set for Gunning Hammock (a), and Cockle Island (a and b) during the depopulation and re-colonization phases. For Gunning Hammock, last depopulation and re-colonization phase CPUEs that differ significantly from that of the initial collection are designated by an asterisk, and CPUEs during the re-colonization phase that are significantly different from the final depopulation collection phase CPUE are indicated by open diamonds. For Cockle Island, collection CPUEs during the re-colonization phase that are significantly different from the final depopulation collection phase CPUE are indicated by solid circles. Linear regression analysis formula and slope line are indicated for the final three depopulation sets at Cockle Island. One standard error for CPUE is indicated for all sets, ($p \leq 0.05$).


Fundulus heteroclitus
Gunning Hammock
Cockle Island
Number of Individuals/Eel Pot
250
225
200
175
150
125
100
75
50
25
0
y = (x) -1.95122 + 12.66667
r2 = 0.3378
p = 0.6052
Cockle Island
40
30
20
10
0
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25
1 2 3 4
Depopulation Set
Re-colonization Set
Collection Phase

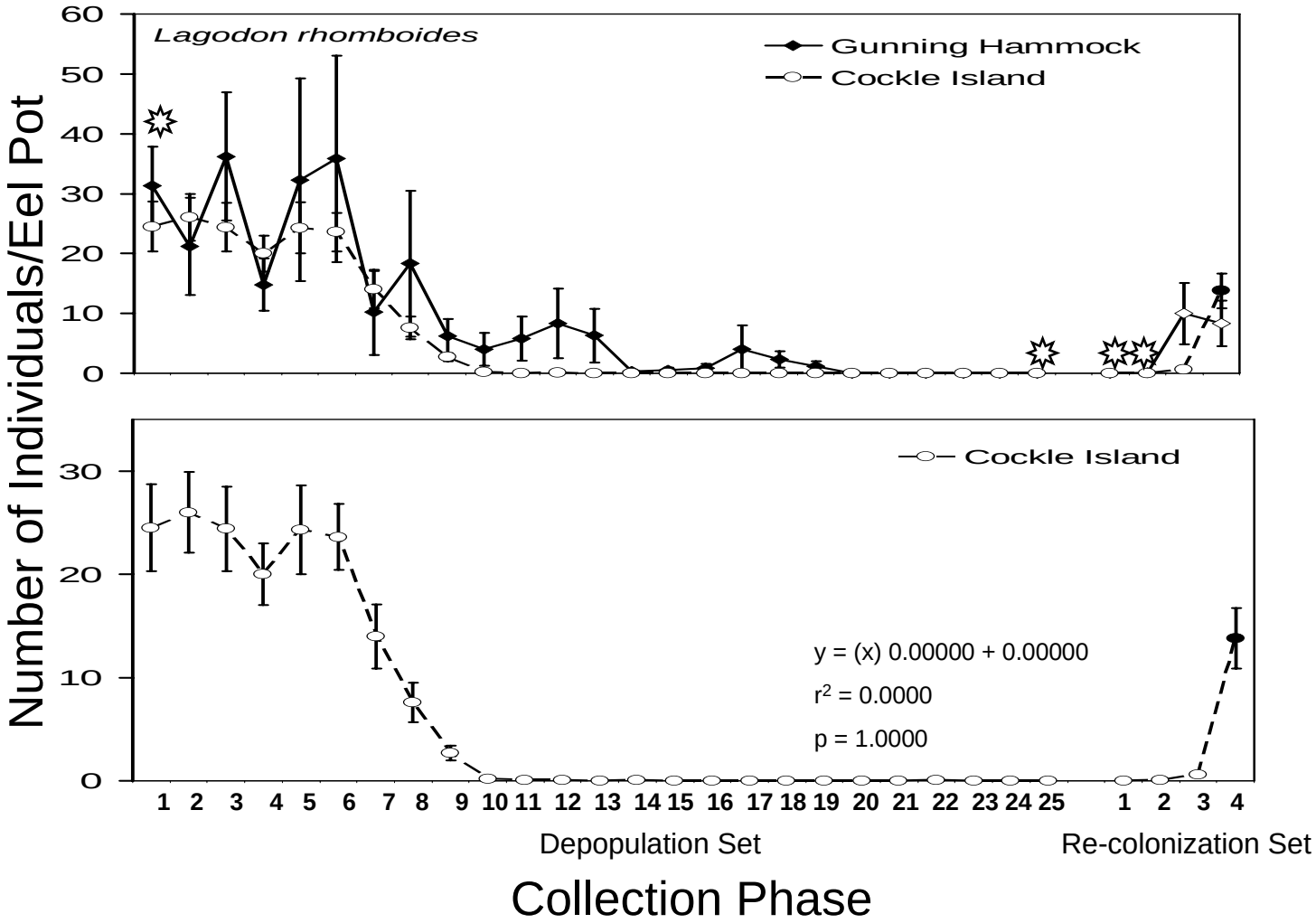
Figs. 4-6a,b. Mean number of Lagodon rhomboides collected per eel pot (CPUE), per set for Gunning Hammock (a), and Horse Island (a and b) during the depopulation and re-colonization phases. For Horse Island, collection CPUEs during the re-colonization phase that are significantly different from the final depopulation phase collection CPUE are indicated by solid circles. Linear regression analysis formula and slope line are indicated for the final three depopulation phase sets at Horse Island. One standard error for CPUE is indicated for all sets, ($p \leq 0.05$).


Lagodon rhomboides
Gunning Hammock
Horse Island
Number of Individuals/Eel Pot
60
50
40
30
20
10
0
y = (x) -0.00000 + 0.54902
r2 = 0.0000
p = 1.0; n = 3
1 2 3 4 5 6 7 8
1 2 3 4
Depopulation Set
Re-colonization Set
Collection Phase

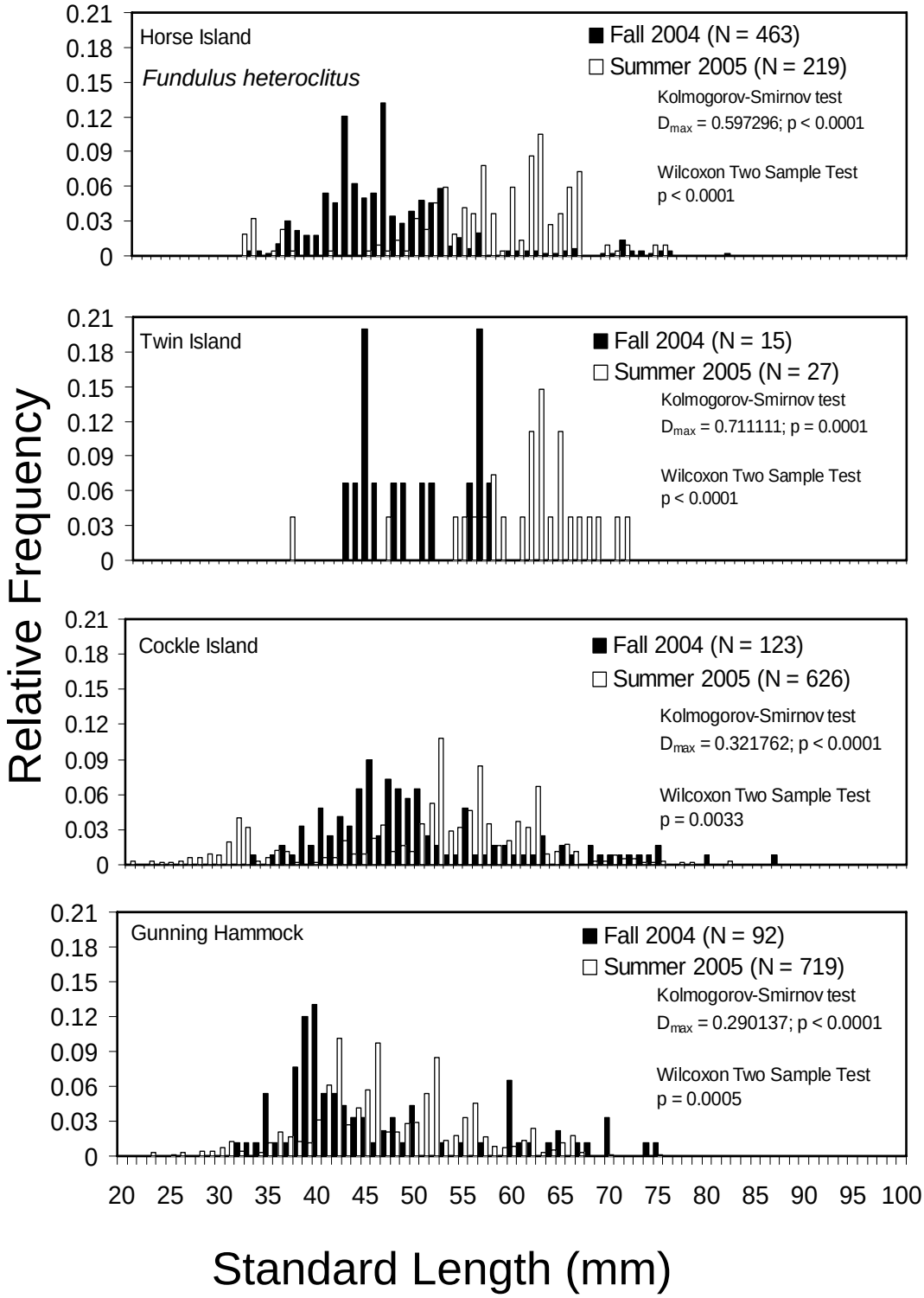
Figs. 4-7a,b. Mean number of Lagodon rhomboides collected per eel pot (CPUE), per set for Gunning Hammock (a), and Twin Island (a and b) during the depopulation and re-colonization phases. For Twin Island, collection CPUEs during the re-colonization phase that are significantly different from the final depopulation collection phase CPUE are indicated by solid circles. Linear regression analysis formula and slope line are indicated for the final three depopulation phase sets at Twin Island. One standard error for CPUE is indicated for all sets, ($p \leq 0.05$).


Lagodon rhomoides
Gunning Hammock
Twin Island
Number of Individuals/Eel Pot
80
70
60
50
40
30
20
10
0
y = (x) -0.04000 + 0.38667
r2 = 0.7500
p = 0.3333; n = 3
1 2 3 4 5 6 7 8 9 10
1 2 3 4
Depopulation Set
Re-colonization Set
Collection Phase

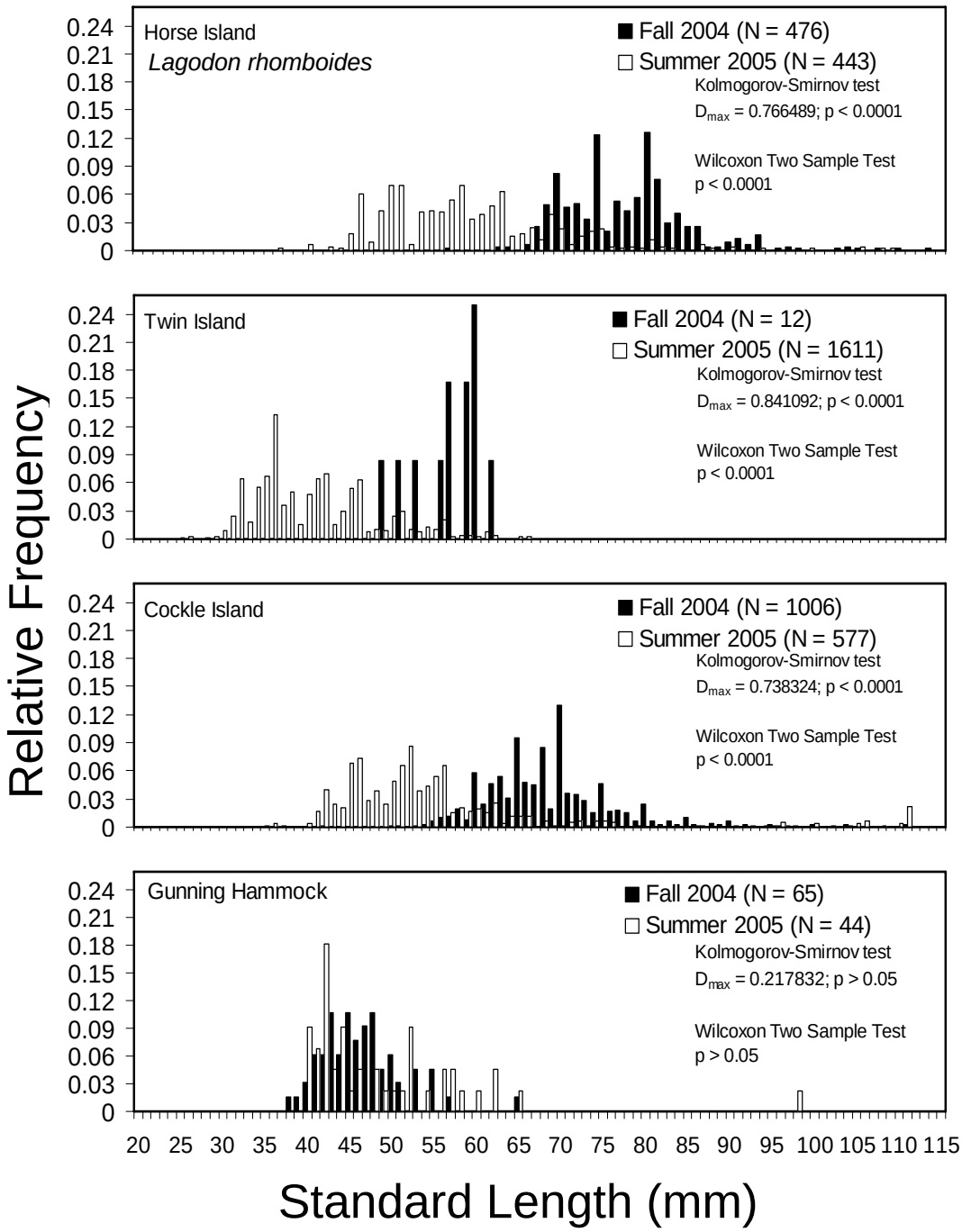
Figs. 4-8a,b. Mean number of Lagodon rhomboides collected per eel pot (CPUE), per set for Gunning Hammock (a), and Cockle Island (a and b) during the depopulation and re-colonization phases. For Gunning Hammock, last depopulation and re-colonization phase CPUEs that differ significantly from that of the initial collection are designated by an asterisk, and CPUEs during the re-colonization phase that are significantly different from the final depopulation collection phase CPUE are indicated by open diamonds. For Cockle Island, collection CPUEs during the re-colonization phase that are significantly different from the final depopulation collection phase CPUE are indicated by solid circles. Linear regression analysis formula and slope line are indicated for the final three depopulation sets at Cockle Island. One standard error for CPUE is indicated for all sets, ($p \leq 0.05$).


Lagodon rhomboides
Gunning Hammock
Cockle Island
Number of Individuals/Eel Pot
60
50
40
30
20
10
0
Cockle Island
y = (x) 0.00000 + 0.00000
r2 = 0.0000
p = 1.0000
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 1 2 3 4
Depopulation Set
Re-colonization Set
Collection Phase

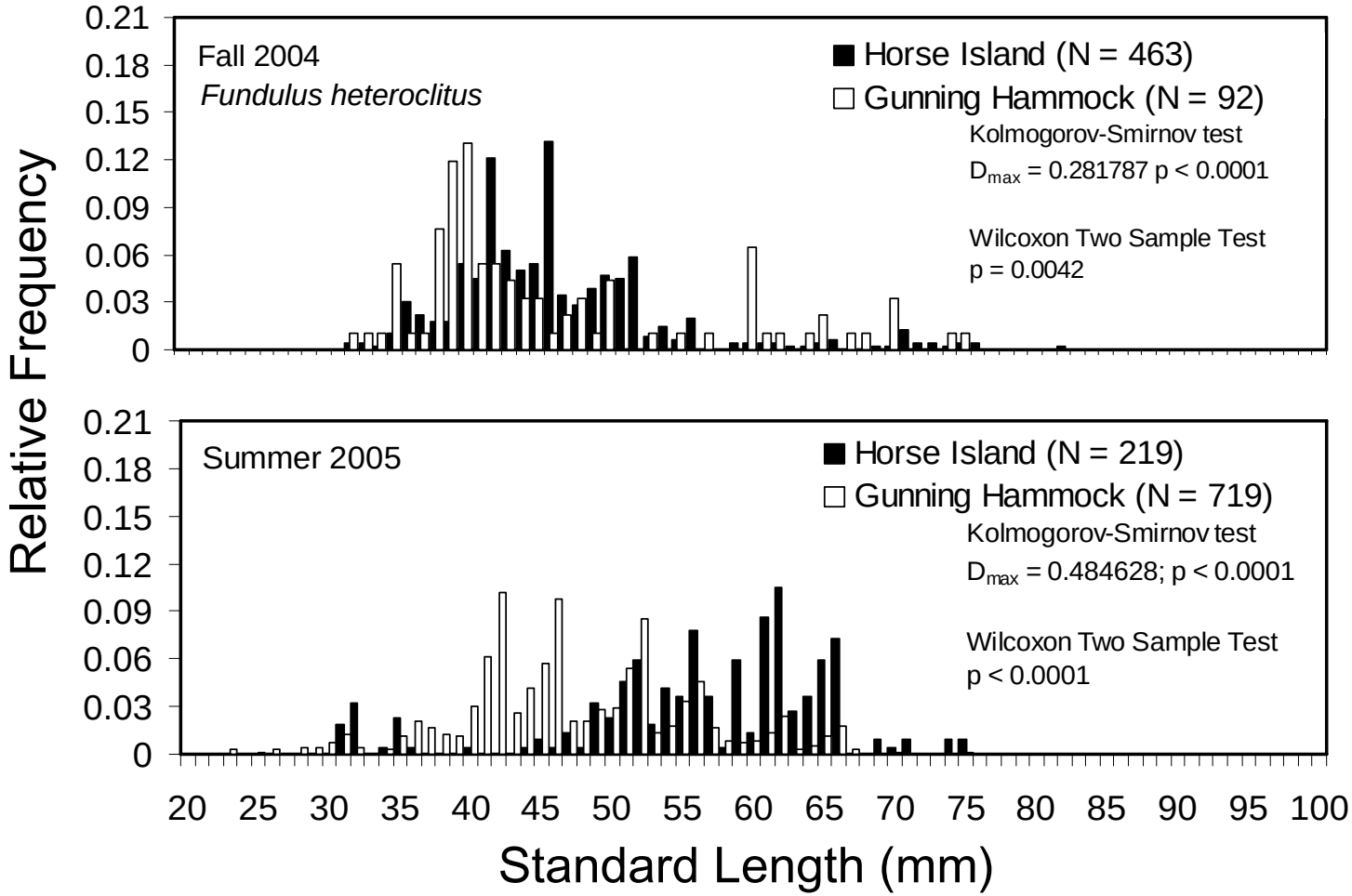
Figs. 4-9a-d. Relative frequency distributions of Fundulus heteroclitus size classes for site comparisons between individuals collected during the initial (fall 2004, high for Twin Island) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Horse Island
Fundulus heteroclitus
Fall 2004 (N = 463)
Summer 2005 (N = 219)
Kolmogorov-Smirnov test
Dmax = 0.597296; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Twin Island
Fall 2004 (N = 15)
Summer 2005 (N = 27)
Kolmogorov-Smirnov test
Dmax = 0.711111; p = 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Cockle Island
Fall 2004 (N = 123)
Summer 2005 (N = 626)
Kolmogorov-Smirnov test
Dmax = 0.321762; p < 0.0001
Wilcoxon Two Sample Test
p = 0.0033
Gunning Hammock
Fall 2004 (N = 92)
Summer 2005 (N = 719)
Kolmogorov-Smirnov test
Dmax = 0.290137; p < 0.0001
Wilcoxon Two Sample Test
p = 0.0005
Relative Frequency
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100
Standard Length (mm)

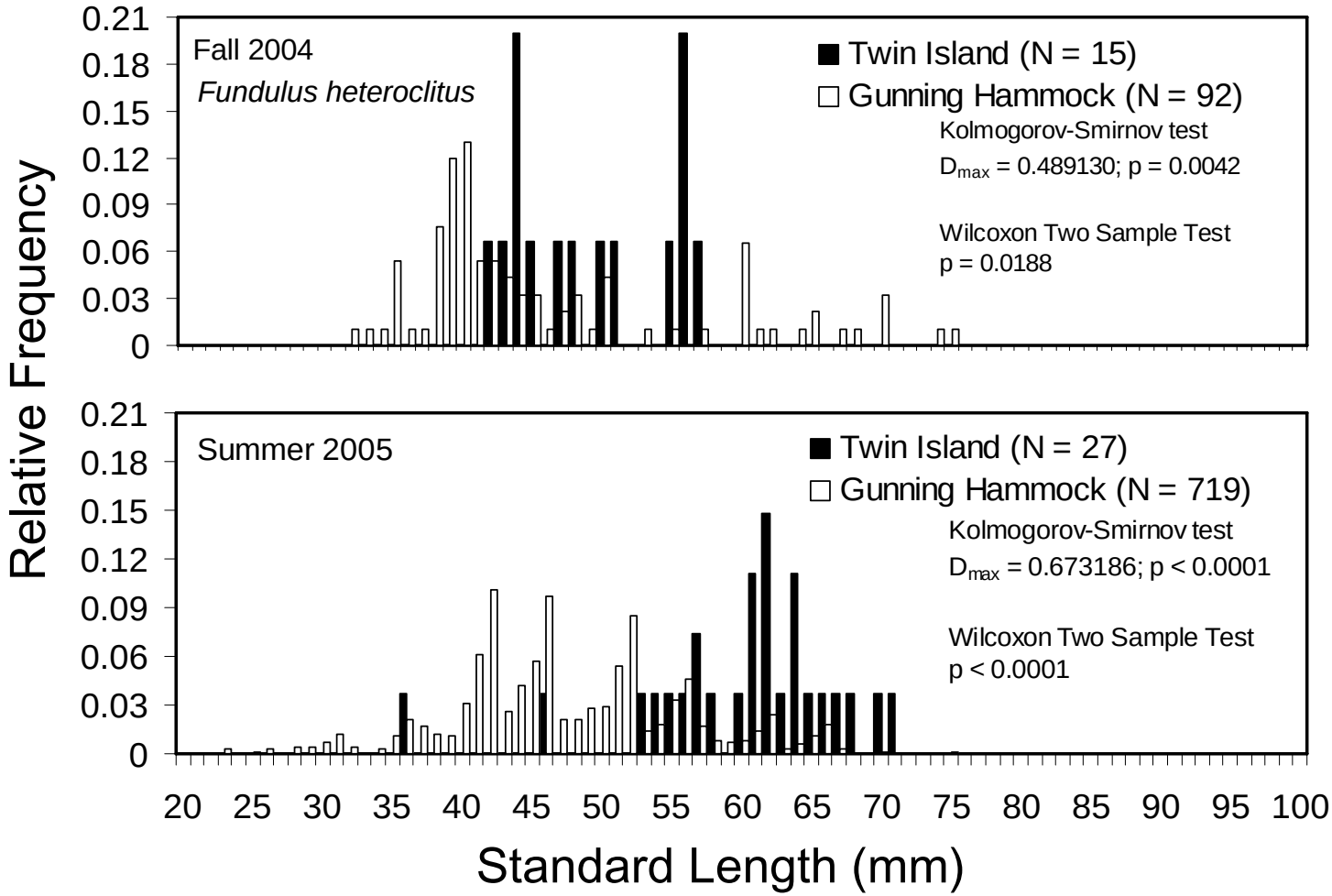
Figs. 4-10a-d. Relative frequency distributions of Lagodon rhomboides size classes for site comparisons between individuals collected during the initial (fall 2004, high for Twin Island) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Horse Island
Lagodon rhomboides
Fall 2004 (N = 476)
Summer 2005 (N = 443)
Kolmogorov-Smirnov test
Dmax = 0.766489; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Twin Island
Fall 2004 (N = 12)
Summer 2005 (N = 1611)
Kolmogorov-Smirnov test
Dmax = 0.841092; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Cockle Island
Fall 2004 (N = 1006)
Summer 2005 (N = 577)
Kolmogorov-Smirnov test
Dmax = 0.738324; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Gunning Hammock
Fall 2004 (N = 65)
Summer 2005 (N = 44)
Kolmogorov-Smirnov test
Dmax = 0.217832; p > 0.05
Wilcoxon Two Sample Test
p > 0.05
Relative Frequency
0.24
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100 105 110 115
Standard Length (mm)

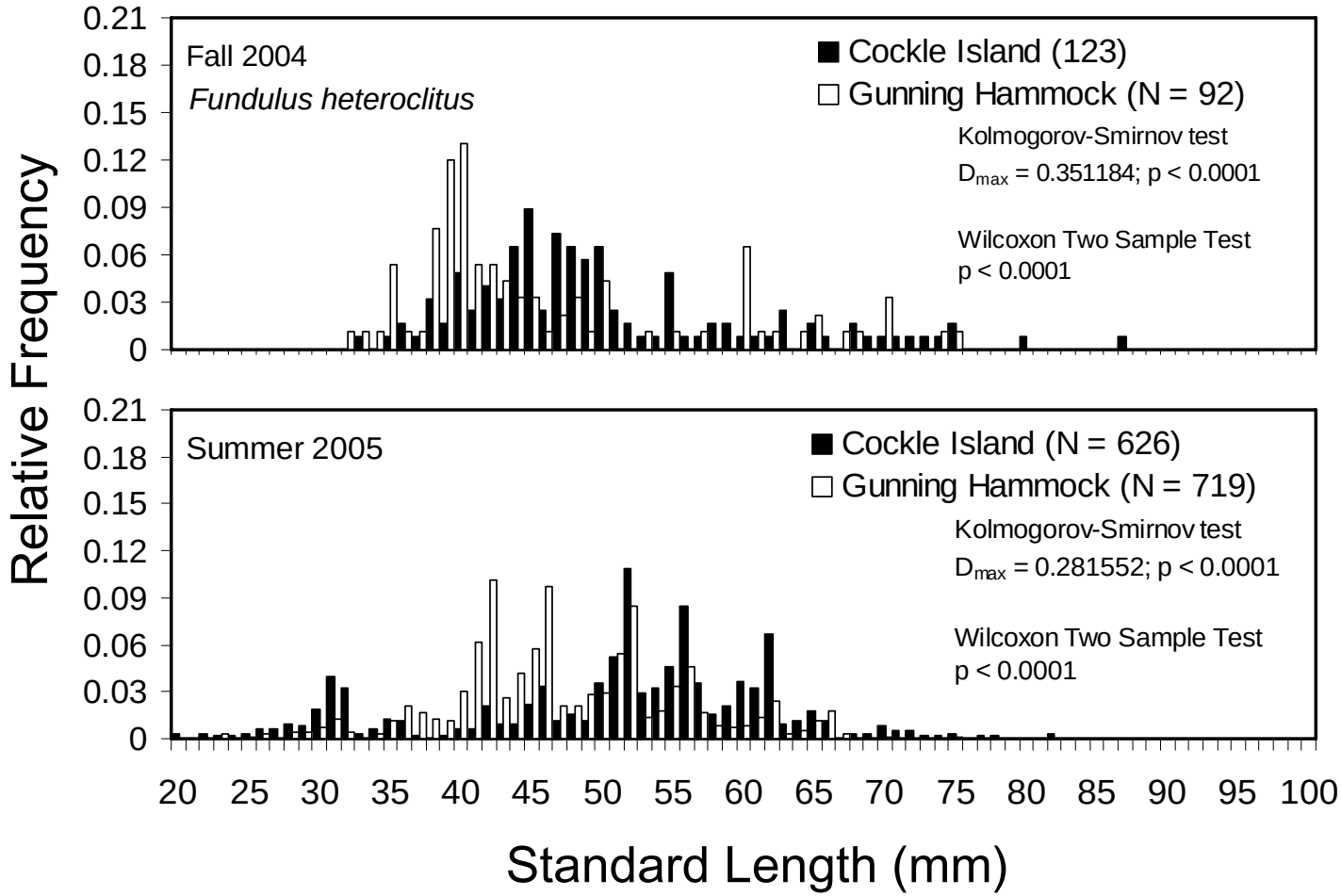
Figs. 4-11a,b. Relative frequency distributions of Fundulus heteroclitus size classes for site comparisons between individuals collected at Horse Island and Gunning Hammock during the initial (fall 2004) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Relative Frequency
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
Fall 2004
Fundulus heteroclitus
Horse Island (N = 463)
Gunning Hammock (N = 92)
Kolmogorov-Smirnov test
Dmax = 0.281787 p < 0.0001
Wilcoxon Two Sample Test
p = 0.0042
Summer 2005
Horse Island (N = 219)
Gunning Hammock (N = 719)
Kolmogorov-Smirnov test
Dmax = 0.484628; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100
Standard Length (mm)

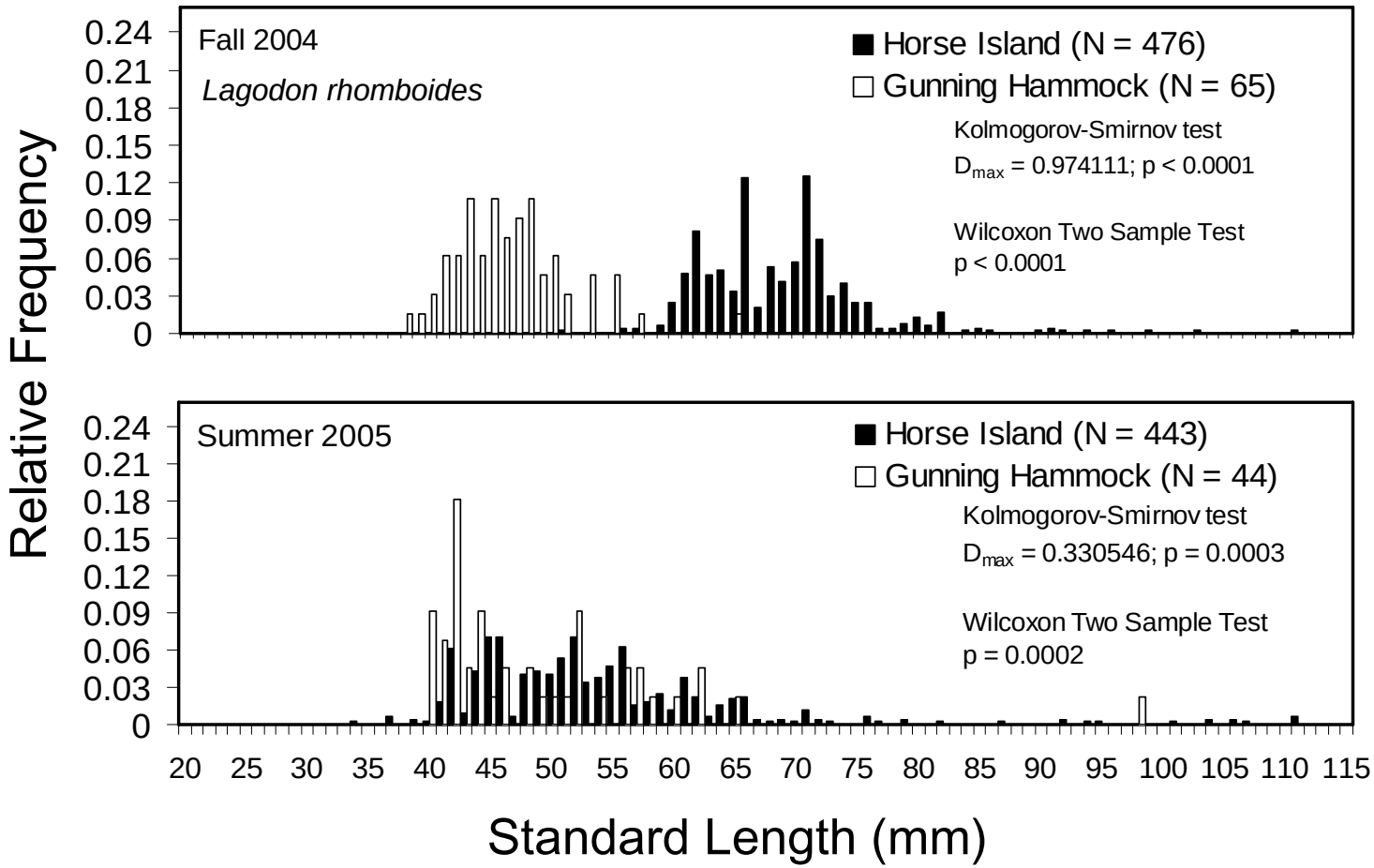
Figs 4-12a,b. Relative frequency distributions of Fundulus heteroclitus size classes for site comparisons between individuals collected at Twin Island and Gunning Hammock during the initial (fall 2004, high for Twin Island) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Relative Frequency
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
Fall 2004
Fundulus heteroclitus
Twin Island (N = 15)
Gunning Hammock (N = 92)
Kolmogorov-Smirnov test
Dmax = 0.489130; p = 0.0042
Wilcoxon Two Sample Test
p = 0.0188
Summer 2005
Twin Island (N = 27)
Gunning Hammock (N = 719)
Kolmogorov-Smirnov test
Dmax = 0.673186; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100
Standard Length (mm)

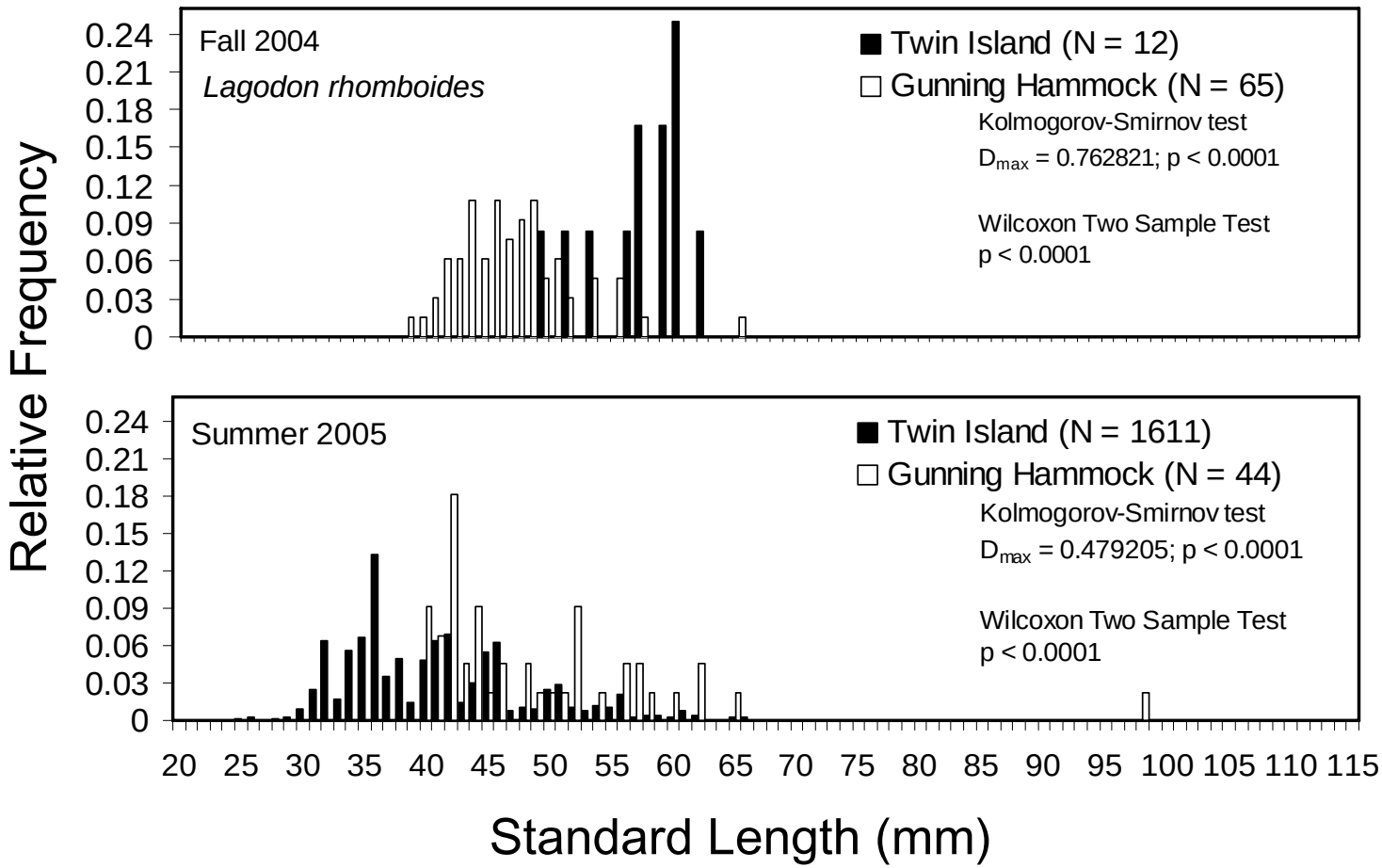
Figs. 4-13a,b. Relative frequency distributions of Fundulus heteroclitus size classes for site comparisons between individuals collected at Cockle Island and Gunning Hammock during the initial (fall 2004) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Relative Frequency
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
Fall 2004
Fundulus heteroclitus
Cockle Island (123)
Gunning Hammock (N = 92)
Kolmogorov-Smirnov test
Dmax = 0.351184; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Summer 2005
Cockle Island (N = 626)
Gunning Hammock (N = 719)
Kolmogorov-Smirnov test
Dmax = 0.281552; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100
Standard Length (mm)

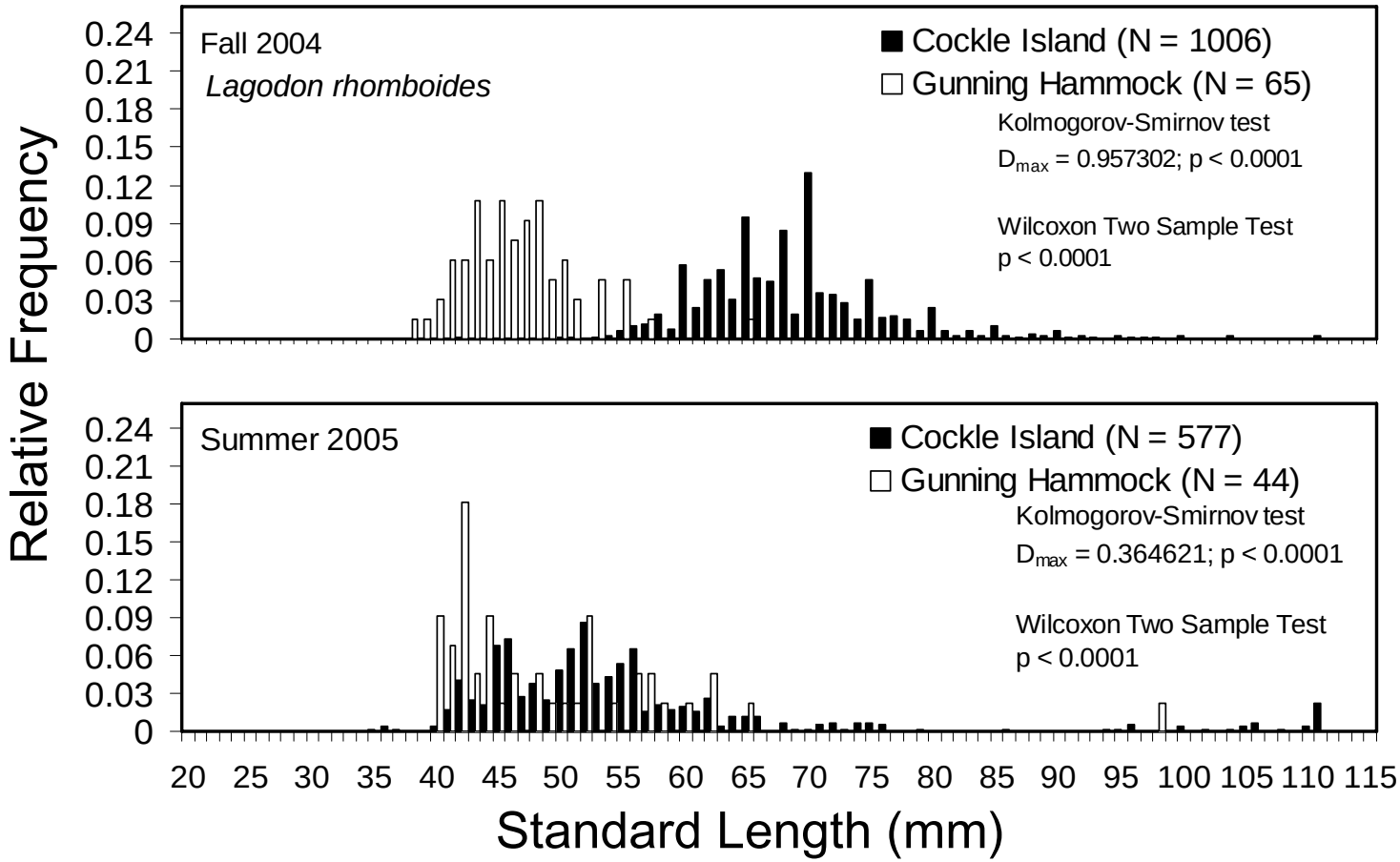
Figs. 4-14a,b. Relative frequency distributions of Lagodon rhomboides size classes for site comparisons between individuals collected at Horse Island and Gunning Hammock during the initial (fall 2004) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Fall 2004
Lagodon rhomboides
Horse Island (N = 476)
Gunning Hammock (N = 65)
Kolmogorov-Smirnov test
Dmax = 0.974111; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Summer 2005
Horse Island (N = 443)
Gunning Hammock (N = 44)
Kolmogorov-Smirnov test
Dmax = 0.330546; p = 0.0003
Wilcoxon Two Sample Test
p = 0.0002
Relative Frequency
0.24
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100 105 110 115
Standard Length (mm)

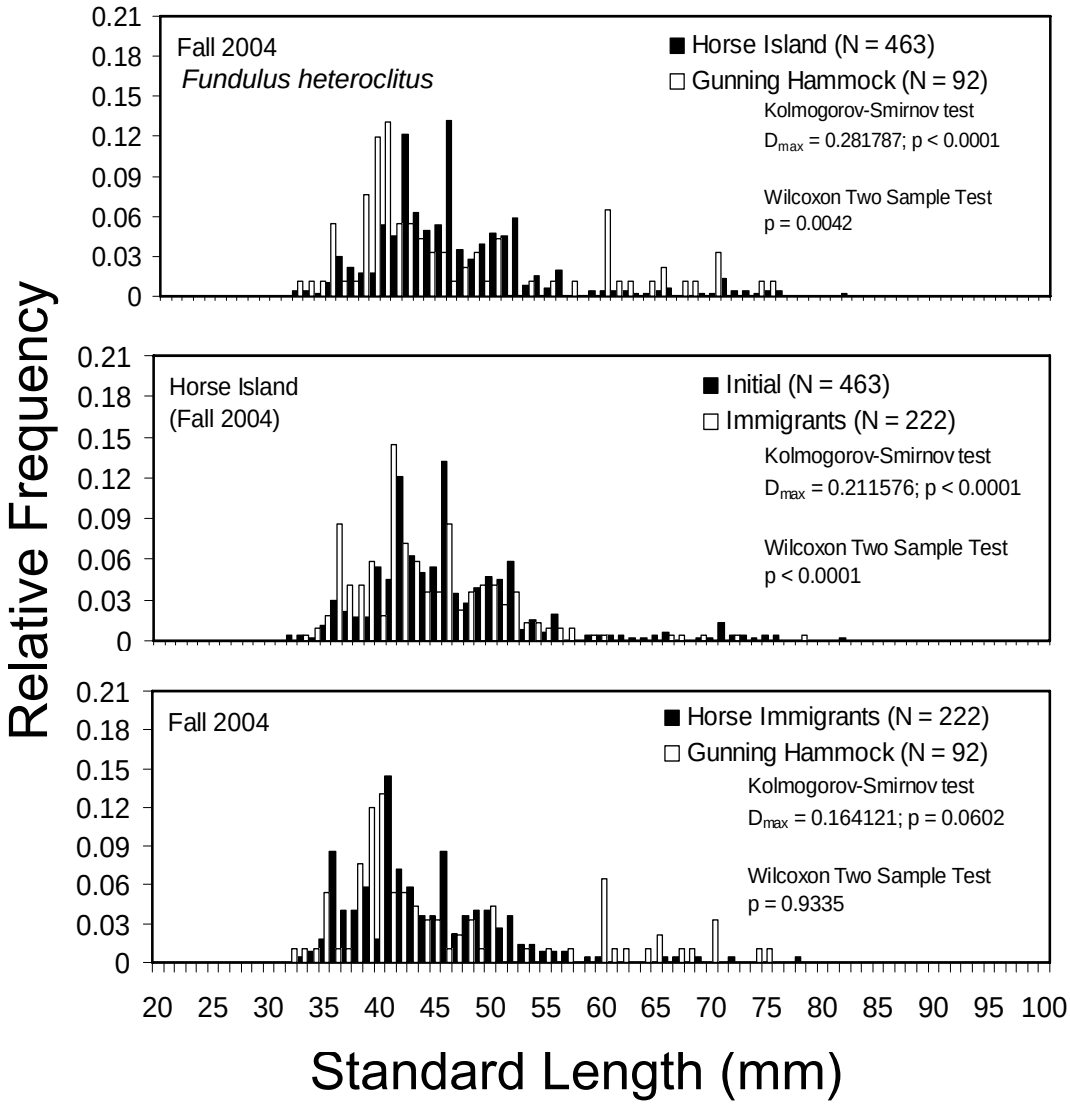
Figs. 4-15a,b. Relative frequency distributions of Lagodon rhomboides size classes for site comparisons between individuals collected at Twin Island and Gunning Hammock during the initial (fall 2004, high for Twin Island) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Fall 2004
Lagodon rhomboides
Twin Island (N = 12)
Gunning Hammock (N = 65)
Kolmogorov-Smirnov test
Dmax = 0.762821; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Summer 2005
Twin Island (N = 1611)
Gunning Hammock (N = 44)
Kolmogorov-Smirnov test
Dmax = 0.479205; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Relative Frequency
0.24
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100 105 110 115
Standard Length (mm)

Figs. 4-16a,b. Relative frequency distributions of Lagodon rhomboides size classes for site comparisons between individuals collected at Cockle Island and Gunning Hammock during the initial (fall 2004) depopulation and final (summer 2005) re-colonization collections. $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Fall 2004
Lagodon rhomboides
Cockle Island (N = 1006)
Gunning Hammock (N = 65)
Kolmogorov-Smirnov test
Dmax = 0.957302; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Summer 2005
Cockle Island (N = 577)
Gunning Hammock (N = 44)
Kolmogorov-Smirnov test
Dmax = 0.364621; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Relative Frequency
0.24
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100 105 110 115
Standard Length (mm)

Figs. 4-17a-c. Relative frequency distributions of Fundulus heteroclitus size classes for site comparisons between individuals collected at: Horse Island and Gunning Hammock during the initial (fall 2004) depopulation collections (a), Horse Island during the initial (fall 2004) depopulation collections and fall 2004 Horse Island immigrants (b), Gunning Hammock during the initial (fall 2004 ) depopulation collections and fall 2004 Horse Island immigrants (c). $D_{max}$ and p values based on Kolmogorov-Smirnov two sample tests for distribution differences, and p values for the Wilcoxon two sample test are represented for mean size comparisons.


Fall 2004
Fundulus heteroclitus
Horse Island (N = 463)
Gunning Hammock (N = 92)
Kolmogorov-Smirnov test
Dmax = 0.281787; p < 0.0001
Wilcoxon Two Sample Test
p = 0.0042
Horse Island
(Fall 2004)
Initial (N = 463)
Immigrants (N = 222)
Kolmogorov-Smirnov test
Dmax = 0.211576; p < 0.0001
Wilcoxon Two Sample Test
p < 0.0001
Fall 2004
Horse Immigrants (N = 222)
Gunning Hammock (N = 92)
Kolmogorov-Smirnov test
Dmax = 0.164121; p = 0.0602
Wilcoxon Two Sample Test
p = 0.9335
Relative Frequency
0.21
0.18
0.15
0.12
0.09
0.06
0.03
0
20 25 30 35 40 45 50 55 60 65 70 75 80 85 90 95 100
Standard Length (mm)

# CHAPTER 5.

# CONCLUSIONS

# CONCLUSIONS

Fundulus heteroclitus exclusively utilized salt marsh habitats during high tide and not adjacent shallow water flats while Lagodon rhomboides utilized both. Fundulus heteroclitus preferred the shallow salt marsh of fringing mainland and island salt marshes over deeper water low salt marsh areas (Chapter 2). A F. heteroclitus distribution gradient has been predicted to occur based on water depth (Ruiz et al. 1993), and data from my study confirms that F. heteroclitus is typically restricted to shallow water refuge regions < 1.0 m deep. Such limitation can also restrict F. heteroclitus usage within salt marsh habitats. By contrast, Lagodon rhomboides was restricted to deeper water areas within salt marshes and were abundant near the salt marsh edge and the shallow water flat habitat, indicating the capability to use a wide range of habitat types within estuaries (Hettler 1989; Meyer et al. 1996; 1999; Wenner et al. 1996; Paperno et al. 2001; Hovel et al. 2002; Chapter 2).

Salt marsh size influenced habitat suitability for estuarine species support and minimal salt marsh area, or sub-habitat size thresholds, exist for certain nekton species (Chapter 2). The size of adjacent habitats and sub-habitats can also influence the population density for rare transient species that occasionally pass through from other habitats (MacArthur and Wilson 1967) or facultatively utilize salt marsh habitats (Eggelston et al. 1998) as well as resident salt marsh species that facultatively utilize adjacent sub-habitats (Fukao 1980). Species spillover between sub-habitats and adjacent habitats (Tewfik and Bene 2003; Zeller et al. 2003) can also influence species occurrence.

Mainland salt marshes occupy one end of an estuarine spectrum and act as refugia for marsh dependent nekton, while small island salt marshes act as refugia for open water

nekton species that spill over into this habitat type from surrounding shallow water flats (Chapter 2). Large island salt marshes provide a transitional intermediate habitat for both estuarine open water and salt marsh dependent nekton (With and Crist 1995) with distribution shifts based on habitat fragmentation effects (Chapter 2).

The density of salt marsh-dependent species, including Fundulus heteroclitus, were an order of magnitude lower within the large island compared to the mainland salt marshes, and essentially nonexistent within the small island salt marshes (Chapter 2). Small salt marsh islands were apparently below the 'extinction threshold', the minimum size of suitable habitat necessary for population persistence (Andren 1994; With and Crist 1995) for this species and other salt marsh dependent species.

Similarities among mainland and isolated large island salt marsh habitats for larvae and subsequent disparity in juvenile Fundulus heteroclitus abundances suggest that a mortality related bottleneck constrained adult and juvenile F. heteroclitus populations at large island salt marshes (Chapter 3). Further, these patterns suggest that population limitation for F. heteroclitus at the large island salt marsh habitats occurs with juvenile sized individuals, restricting subsequent adult F. heteroclitus populations. Differential predation rates among the different salt marsh types might have significantly contributed to disparities in young of year abundance (Chapter 3).

Predator concentrations were consistently higher during peak spawning and recruitment season for F. heteroclitus and during times of peak adult and juvenile F. heteroclitus occurrence (Chapter 3). Predation pressure increase would be greatest within the island salt marsh habitats, due partly to higher predator concentrations prevalent at

these sites, and greater coverage potential of this salt marsh habitat compared to mainland salt marsh. Predation would also disrupt migration between separate salt marsh habitats, effectively isolating marsh dependant species such as F. heteroclitus within geographically separate salt marshes. Hence, island salt marshes probably become most isolated from other salt marshes during the crucial breeding season. Isolation during critical periods, including breeding season, would contribute to decreased population maintenance potential and increased susceptibility of extinction. Further, significant differences in the area/perimeter and area-perimeter/predator ratios among the salt marsh types implies that island salt marsh sub-habitats had greater predator accessability from multiple directions and coverage potential on a unit area basis compared to mainland salt marsh habitat (Chapter 3). This would reduce predation refuge quality for island salt marshes and significantly increase predation related mortality for salt marsh dependant nekton compared to mainland salt marshes.

Movement corridors (including shallow water flats, banks and bars utilized during low tide) between salt marshes for Fundulus heteroclitus are suspected to open annually during the late fall through early spring time periods when predator concentrations are at an annual low (Chapter 3), and tides are astronomically low (Hutchinson and Sklar 1993). These movement corridors essentially close during other times of the year when predation threat significantly increases.

Immigration to the islands happened quickly for Fundulus heteroclitus, within weeks or months of depopulation efforts. The rapid re-establishment of a F. heteroclitus population at Horse Island indicated that dispersive movement was occurring during the

depopulation time period (Chapter 4). The rapid re-establishment of a F. heteroclitus population at Horse Island indicated that not only did F. heteroclitus immigrate to isolated island salt marshes across substantial open water distances (at least 800 m), but that movement of large population numbers can occur in a relatively rapid time period. For Lagodon rhomboides, immigration to the islands and mainland was not detected until the population had reached sufficient size classes targeted for the collection gear, during May and July but probably had colonized these habitats by late winter (Warlen and Burke 1990; Meyer et al. 1996; Chapter 2).

There are two crucial time periods for Fundulus heteroclitus population maintenance, one occurs during the late spring and early summer spawn and recruitment period (Talbot and Able 1984; Abrams 1985) and the second during the dispersal period for emigration-sized F. heteroclitus which occurs in the late fall through early spring when water temperatures are colder, predator levels are significantly reduced (Chapter 3) and corridors are available within a suitable water depth range (Chapter 2). The emigration/immigration phase of this population distribution pattern for F. heteroclitus might be the more significant for maintaining the geographic distribution for this species among island salt marshes, while the breeding/recruitment phase is critical for local viable populations in large salt marsh areas. By contrast, for transient species such as Lagodon rhomboides, the crucial mechanism to recruitment and dispersal occur simultaneously during the late fall to late winter time period when spawning, dispersal and recruitment to the various habitats within the coastal estuaries occurs (Warlen and Burke 1990) and low predator levels (Chapter 3) create and optimal time for both

geographical and local population maintenance. Both F. heteroclitus and L. rhomboides are capable of seasonally colonizing salt marsh habitat at the level of isolation investigated during this study.

The availability of sufficient salt marsh habitat with water deep enough to allow resident salt marsh nekton usage, but shallow enough to restrict predator incursion into salt marsh habitat, is thus hypothesized to be a primary factor contributing to the consistent density patterns observed for resident marsh nekton between salt marsh types. The amount of shallow salt marsh habitat at the islands sampled during this study was also identified as the leading factor among numerous biotic, physical, environmental and habitat factors examined, to directly relate resident salt marsh nekton, and particularly F. heteroclitus, density to a predictable model (Chapter 2). Similarly, comparisons among salt marsh habitat based on size categories (mainland, large and small island) examined also indicated an increase in density for resident marsh nekton including F. heteroclitus, Fundulus luciae, Cyprinodon variegatus and Lucania parva that directly followed the estimated shallow salt marsh refuge area (minimal estimates for mainland) (Chapter 2). For island salt marshes that are isolated from like habitat, shallow salt marsh habitat is critical for maintaining resident salt marsh nekton populations.

Because size and location do matter, future salt marsh preservation and restoration efforts should be based on a strategy that might increase potential overall success pertaining to the use by resident marsh nekton. This could include the clustering of preserved or restored salt marshes within a specific area, or placement of restored salt marsh near other existing salt marshes linked by movement corridor habitats including

shallow water banks, bars or flats to increase occupancy potential by resident species and resident species density (Dunning et al. 1995). This would not only allow utilization by transient species, but also that of resident species with limited dispersal ability. Differences in usage patterns by nekton based on location and habitat size also suggests that preservation and restoration efforts can target specific species or groups of species for management efforts. Habitat components of salt marshes can be adjusted to produce population density targets for certain resident salt marsh species.

When considering future habitat management issues it is necessary that landscape ecology, metapopulation, island biogeography, patch dynamic and migration corridor theory be considered. By using theoretical models combined with complimentary empirical studies, the functional aspects of estuarine habitats such as salt marsh habitat can be better understood and estuarine ecosystems managed for future generations.

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# APPENDICES.

Appendix A. Total number of all nekton species by time period and for the year, collected for each marsh type via block and fyke net collections.

| | Date | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Treatment Type | 6/02 | 11/02 | 3/03 | Total Year One | 6/03 | 11/03 | 3/04 | Total Year Two |
| Mainland Salt Marsh | 23 | 23 | 19 | 35 | 20 | 23 | 16 | 31 |
| Large Island Salt Marsh | 20 | 18 | 16 | 31 | 18 | 15 | 17 | 30 |
| Small Island Salt Marsh | 15 | 12 | 12 | 25 | 14 | 13 | 13 | 24 |

Appendix B. Number of individuals 10 $m^{-2}$ (cumulative for total years 1 and 2), for the fish and decapod species and the species that made up the top 95% of all individuals collected. For each species during each collection date, comparisons between large island marsh (LIM) and small island marsh (SIM) treatments that are significantly different from one another at $p \leq 0.05$ are indicated by a different letter. Numbers in parentheses are one standard error.

| Date | ------June 2002------- | | ----November 2002---- | | ------March 2003------ | | ------Total Year ------ | |
|---|---|---|---|---|---|---|---|---|
| Species | LIM | SIM | LIM | SIM | LIM | SIM | LIM | SIM |
| Cyprinodon variegatus | 0.59(±0.39) | 0.00(±0.00) | 0.03(±0.02) | 0.00(±0.00) | 0.06(±0.02)$^{B}$ | 0.24(±0.08)$^{A}$ | 0.68(±0.41) | 0.24(±0.08) |
| Eucinostomus spp. | 0.02(±0.02) | 0.00(±0.00) | 3.91(±3.82) | 0.42(±0.17) | 0.00(±0.00) | 0.00(±0.00) | 3.93(±3.82) | 0.42(±0.17) |
| Fundulus heteroclitus | 1.77(±0.90) | 0.17(±0.12) | 0.30(±0.12) | 0.06(±0.06) | 1.02(±0.72) | 0.06(±0.03) | 3. 09(±1.54)$^{A}$ | 0.29(±0.11)$^{B}$ |
| Fundulus lucia | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus majalis | 2.31(±0.34) | 8.76(±4.55) | 0.46(±0.36) | 0.14(±0.14) | 1.16(±1.10) | 15.93(±11.48) | 3.93(±1.75) | 24.83(±15.72) |
| Gobionellus boleosoma | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Gobiosoma bosc | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Lagodon rhomboides | 25.76(±7.72) | 15.52(±10.60) | 5.56(±5.17) | 2.49(±1.07) | 10.41(±8.46) | 2.35(±1.01) | 41.73(±8.77) | 20.36(±11.98) |
| Leiostomus xanthurus | 0.38(±0.32) | 0.91(±0.20) | 0.14(±0.14) | 0.14(±0.14) | 906.16(±878.33) | 51.55(±9.03) | 906.68(±878.59) | 52.60(±9.34) |
| Lucania parva | 0.06(±0.06) | 0.00(±0.00) | 0.44(±0.44) | 0.00(±0.00) | 0.53(±0.53) | 0.00(±0.00) | 1.03(±0.53) | 0.00(±0.00) |
| Menidia menidia | 0.00(±0.00)$^{B}$ | 70.57(±46.92)$^{A}$ | 8.76(±8.52) | 21.05(±13.05) | 5.54(±5.36) | 3.88(±2.90) | 14.30(±7.35) | 95.50(±58.82) |
| Mugil cephalus | 0.00(±0.00) | 0.00(±0.00) | 0.05(±0.05) | 0.00(±0.00) | 0.28(±0.04)$^{A}$ | 0.00(±0.00)$^{B}$ | 0.33(±0.01)$^{A}$ | 0.00(±0.00)$^{B}$ |
| Mugil curema | 0.02(±0.02)$^{B}$ | 0.59(±0.24)$^{A}$ | 0.05(±0.05) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.07(±0.05)$^{B}$ | 0.59(±0.24)$^{A}$ |
| Paralichthys albigutta | 0.31(±0.01) | 0.40(±0.08) | 0.11(±0.11) | 0.00(±0.00) | 0.00(±0.00) | 0.03(±0.03) | 0.42(±0.10) | 0.43(±0.07) |
| Paralichthys dentatus | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.65(±0.53) | 0.43(±0.28) | 0.65(±0.53) | 0.43(±0.28) |
| Other Fishes | 0.26(±0.14) | 0.17(±0.04) | 0.09(±0.05) | 0.03(±0.03) | 0.08(±0.05) | 0.11(±0.07) | 0.43(±0.14) | 0.31(±0.11) |
| Callinectes sapidus | 2.99(±0.43) | 1.86(±0.97) | 0.12(±0.12) | 1.33(±0.90) | 0.08(±0.05) | 0.00(±0.00) | 3.19(±0.48) | 3.19(±0.86) |
| Callinectes similis | 0.00(±0.00) | 2.06(±2.06) | 0.00(±0.00) | 0.04(±0.04) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 2.10(±2.09) |
| Clibanarius vittatus | 0.02(±0.02) | 0.07(±0.07) | 0.00(±0.00) | 0.18(±0.12) | 0.04(±0.04) | 0.00(±0.00) | 0.06(±0.04) | 0.25(±0.13) |
| Palaemonetes intermedius | 0.18(±0.18) | 0.00(±0.00) | 1.34(±1.34) | 1.55(±0.59) | 7.71(±0.75) | 0.00 (±0.00) | 9.23(±6.80) | 1.55(±0.59) |
| Palaemonetes pugio | 0.28(±0.22) | 0.00(±0.00) | 0.19(±0.19) | 0.40(±0.40) | 0.28(±0.21) | 0.03(±0.03) | 0.75(±0.41) | 0.43(±0.39) |
| Palaemonetes vulgaris | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.55(±0.28) | 0.10(±0.10) | 0.55(±0.28) | 0.10(±0.10) |
| Farfantepenaeus aztecus | 0.00(±0.00) | 0.18(±0.18) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.18(±0.18) |
| Farfantepenaeus duorarum | 0.00(±0.00) | 0.00(±0.00) | 0.04(±0.02)$^{B}$ | 0.24(±0.08)$^{A}$ | 0.00(±0.00) | 0.00(±0.00) | 0.04(±0.02)$^{B}$ | 0.24(±0.08)$^{A}$ |
| Other Decapods | 0.48(±0.36) | 0.03(±0.03) | 0.05(±0.05) | 0.31(±0.11) | 0.31(±0.20) | 0.09(±0.09) | 0.84(±0.45) | 0.43(±0.05) |

Appendix B. (Continued).

| Date | ---------June 2003--------- | | ----November 2003---- | | ------March 2004------ | | ------Total Year ------- | |
|---|---|---|---|---|---|---|---|---|
| Species | LIM | SIM | LIM | SIM | LIM | SIM | LIM | SIM |
| Cyprinodon variegatus | 0.06(±0.06) | 0.00(±0.00) | 10.73(±5.80) | 0.00(±0.00) | 0.03(±0.03)$^{B}$ | 0.16(±0.05)$^{A}$ | 10.82(±5.82) | 0.16(±0.05) |
| Eucinostomus spp. | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus heteroclitus | 0.59(±0.30) | 0.35(±0.24) | 1.89(±1.32)$^{A}$ | 0.00(±0.00)$^{B}$ | 0.05(±0.03) | 0.51(±0.46) | 2.53(±1.35) | 0.86(±0.70) |
| Fundulus lucia | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Fundulus majalis | 0.08(±0.05) | 40.60(±39.23) | 3.11(±2.09) | 1.87(±1.82) | 1.48(±0.91) | 2.71(±0.78) | 4.67(±2.96) | 45.18(±38.30) |
| Gobionellus boleosoma | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) |
| Gobiosoma bosc | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) |
| Lagodon rhomboides | 15.61(±0.75) | 36.23(±16.17) | 0.23(±0.14) | 6.15(±3.76) | 20.94(±16.74) | 8.48(±7.74) | 36.78(±17.61) | 50.86(±25.09) |
| Leiostomus xanthurus | 1.67(±0.93) | 3.32(±0.89) | 0.00(±0.00) | 0.72(±0.72) | 122.50(±66.80) | 11.66(±2.10) | 124.17(±66.70) | 15.70(±2.33) |
| Lucania parva | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) |
| Menidia menidia | 0.24(±0.16) | 30.99(±29.72) | 0.60(±0.30) | 0.78(±0.58) | 0.44(±0.09) | 0.27(±0.27) | 1.28(±0.42)$^{B}$ | 32.04(±29.59)$^{A}$ |
| Mugil cephalus | 0.52(±0.52) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.62(±0.04)$^{A}$ | 0.00(±0.00)$^{B}$ | 1.14(±0.55)$^{A}$ | 0.00(±0.00)$^{B}$ |
| Mugil curema | 0.00(±0.00) | 0.18(±0.09) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.18(±0.09) |
| Paralichthys albigutta | 0.24(±0.14) | 0.11(±0.07) | 0.04(±0.04) | 0.08(±0.08) | 0.00(±0.00) | 0.00(±0.00) | 0.28(±0.15) | 0.19(±0.15) |
| Paralichthys dentatus | 0.14(±0.04) | 0.35(±0.30) | 0.00(±0.00) | 0.06(±0.03) | 0.66(±0.23) | 0.42(±0.05) | 0.80(±0.26) | 0.83(±0.24) |
| Other Fishes | 0.95(±0.71) | 0.82(±0.82) | 0.02(±0.02)$^{B}$ | 0.13(±0.02)$^{A}$ | 0.02(±0.02) | 0.07(±0.04) | 0.99(±0.72) | 1.02(±0.82) |
| Callinectes sapidus | 3.05(±0.47) | 5.95(±1.10) | 0.21(±0.11)$^{B}$ | 1.81(±0.23)$^{A}$ | 0.35(±0.15)$^{A}$ | 0.00(±0.00)$^{B}$ | 3.61(±0.59)$^{B}$ | 7.76(±1.25)$^{A}$ |
| Callinectes similis | 0.04(±0.04) | 0.38(±0.38) | 0.00(±0.00) | 0.20(±0.20) | 0.00(±0.00) | 0.00(±0.00) | 0.04(±0.04) | 0.58(±0.33) |
| Clibanarius vittatus | 2.91(±1.46) | 2.79(±1.66) | 0.00(±0.00) | 0.36(±0.36) | 0.00(±0.00) | 0.00(±0.00) | 2.91(±1.46) | 3.15(±1.99) |
| Palaemonetes intermedius | 0.18(±0.10) | 0.06(±0.06) | 0.04(±0.04) | 0.00(±0.00) | 0.71(±0.24) | 0.14(±0.14) | 0.93(±0.17)$^{A}$ | 0.20(±0.12)$^{B}$ |
| Palaemonetes pugio | 0.00(±0.00) | 0.17(±0.07) | 1.42(±1.36) | 0.00(±0.00) | 1.80(±0.87)$^{A}$ | 0.03(±0.03)$^{B}$ | 3.22(±2.15)$^{A}$ | 0.20(±0.11)$^{B}$ |
| Palaemonetes vulgaris | 0.00(±0.00) | 0.00(±0.00) | 0.06(±0.06) | 0.00(±0.00) | 2.06(±0.93)$^{A}$ | 0.00(±0.00)$^{B}$ | 2.12(±0.98)$^{A}$ | 0.00(±0.00)$^{B}$ |
| Farfantepenaeus aztecus | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) | 0.02(±0.02) | 0.00(±0.00) |
| Farfantepenaeus duorarum | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.11(±0.07) | 0.00(±0.00) | 0.00(±0.00) | 0.02(±0.02) | 0.11(±0.07) |
| Other Decapods | 0.16(±0.05)$^{A}$ | 0.00(±0.00)$^{B}$ | 0.04(±0.04) | 0.31(±0.26) | 0.13(±0.05) | 0.47(±0.43) | 0.33(±0.03) | 0.78(±0.39) |

Appendix C. Rank of fish and decapod species observed in fyke and block net collections from mainland, large island and small island salt marsh based on percent abundance and biomass.

Year One

ABUNDANCE

| Mainland | | Large Island | | Small Island | |
|---|---|---|---|---|---|
| RANK SPECIES | % | RANK SPECIES | % | RANK SPECIES | % |
| 1 Leiostomus xanthurus | 27.7 | 1 Leiostomus xanthurus | 91.1 | 1 Menidia menidia | 44.6 |
| 2 Lagodon rhomboides | 16.3 | 2 Lagodon rhomboides | 4.3 | 2 Leiostomus xanthurus | 26.1 |
| 3 Fundulus heteroclitus | 11.7 | 3 Menidia menidia | 1.6 | 3 Fundulus majalis | 13.4 |
| 4 Menidia menidia | 7.8 | 4 Palaemonetes intermedius | 0.9 | 4 Lagodon rhomboides | 10.2 |
| 5 Lucania parva | 6.9 | 5 Eucinostomus spp. | 0.5 | 5 Callinectes sapidus | 1.6 |
| 6 Cyprinodon variegatus | 5.3 | 6 Fundulus majalis | 0.4 | 6 Callinectes similis | 1.2 |
| 6 Eucinostomus spp. | 5.3 | 7 Fundulus heteroclitus | 0.3 | 7 Palaemonetes intermedius | 0.8 |
| 7 Palaemonetes pugio | 4.1 | 7 Callinectes sapidus | 0.3 | 8 Other Nekton | 0.6 |
| 8 Palaemonetes intermedius | 3.9 | 8 Other Nekton | 0.2 | 9 Mugil curema | 0.3 |
| 9 Mugil cephalus | 3.6 | 9 Paralichthys dentatus | 0.1 | 10 Eucinostomus spp | 0.2 |
| 10 Fundulus majalis | 2.4 | 9 Cyprinodon variegatus | 0.1 | 10 Fundulus heteroclitus | 0.2 |
| 11 Other Nekton | 1.6 | 9 Lucania parva | 0.1 | 10 Paralichthys albigutta | 0.2 |
| 12 Callinectes sapidus | 1.4 | 9 Palaemonetes pugio | 0.1 | 10 Palaemonetes pugio | 0.2 |
| 13 Mugil curema | 0.8 | 10 Mugil cephalus | <0.1 | 10 Paralichthys dentatus | 0.2 |
| 14 Paralichthys dentatus | 0.4 | 10 Mugil curema | <0.1 | 11 Cyprinodon variegatus | 0.1 |
| 15 Gobiosoma bosc | 0.3 | 10 Paralichthys albigutta | <0.1 | 11 Farfantepenaeus aztecus | 0.1 |
| 15 Farfantepenaeus aztecus | 0.3 | 11 Callinectes similis | 0.0 | 12 Gobiosoma bosc | 0.0 |
| 16 Paralichthys albigutta | 0.1 | 11 Gobiosoma bosc | 0.0 | 12 Lucania parva | 0.0 |
| 17 Callinectes similis | <0.1 | 11 Farfantepenaeus aztecus | 0.0 | 12 Mugil cephalus | 0.0 |

BIOMASS

| Mainland | | Large Island | | Small Island | |
|---|---|---|---|---|---|
| RANK SPECIES | % | RANK SPECIES | % | RANK SPECIES | % |
| 1 Lagodon rhomboides | 28.9 | 1 Leiostomus xanthurus | 47.8 | 1 Menidia menidia | 34.8 |
| 2 Fundulus heteroclitus | 13.1 | 2 Lagodon rhomboides | 18.8 | 2 Fundulus majalis | 25.5 |
| 3 Callinectes sapidus | 12.2 | 3 Callinectes sapidus | 12.7 | 3 Lagodon rhomboides | 14.5 |
| 4 Menidia menidia | 10.2 | 4 Menidia menidia | 12.2 | 4 Callinectes sapidus | 13.7 |
| 5 Mugil cephalus | 7.9 | 5 Fundulus majalis | 2.7 | 5 Callinectes similis | 3.3 |
| 6 Fundulus majalis | 7.1 | 6 Fundulus heteroclitus | 2.0 | 6 Leiostomus xanthurus | 3.2 |
| 7 Paralichthys albigutta | 4.2 | 7 Paralichthys albigutta | 1.4 | 7 Paralichthys albigutta | 2.8 |
| 8 Cyprinodon variegatus | 4.1 | 8 Other Nekton | 0.5 | 8 Other Nekton | 0.6 |
| 9 Leiostomus xanthurus | 4.0 | 9 Eucinostomus spp. | 0.4 | 9 Farfantepenaeus aztecus | 0.4 |
| 10 Mugil curema | 2.2 | 9 Palaemonetes intermedius | 0.4 | 10 Eucinostomus spp. | 0.3 |
| 11 Lucania parva | 1.6 | 10 Mugil cephalus | 0.3 | 11 Fundulus heteroclitus | 0.2 |
| 12 Eucinostomus spp. | 1.5 | 11 Cyprinodon variegatus | 0.2 | 11 Mugil curema | 0.2 |
| 13 Other Nekton | 1.0 | 11 Mugil curema | 0.2 | 12 Cyprinodon variegatus | 0.1 |
| 14 Farfantepenaeus aztecus | 0.7 | 12 Lucania parva | 0.1 | 13 Palaemonetes intermedius | <0.1 |
| 15 Palaemonetes intermedius | 0.4 | 12 Paralichthys dentatus | 0.1 | 13 Palaemonetes pugio | <0.1 |
| 15 Palaemonetes pugio | 0.4 | 13 Palaemonetes pugio | <0.1 | 13 Paralichthys dentatus | <0.1 |
| 16 Paralichthys dentatus | 0.2 | 14 Callinectes similis | 0.0 | 14 Gobiosoma bosc | 0.0 |
| 17 Callinectes similis | 0.1 | 14 Gobiosoma bosc | 0.0 | 14 Lucania parva | 0.0 |
| 18 Gobiosoma bosc | <0.1 | 14 Farfantepenaeus aztecus | 0.0 | 14 Mugil cephalus | 0.0 |

Appendix C. (Continued).

Year Two

ABUNDANCE

| Mainland | | | Large Island | | | Small Island | | |
|---|---|---|---|---|---|---|---|---|
| RANK | SPECIES | % | RANK | SPECIES | % | RANK | SPECIES | % |
| 1 | Leiostomus xanthurus | 47.6 | 1 | Leiostomus xanthurus | 64.7 | 1 | Lagodon rhomboides | 33.2 |
| 2 | Lagodon rhomboides | 15.8 | 2 | Lagodon rhomboides | 18.3 | 2 | Fundulus majalis | 24.6 |
| 3 | Fundulus heteroclitus | 8.1 | 3 | Cyprinodon variegatus | 4.9 | 3 | Leiostomus xanthurus | 15.9 |
| 4 | Palaemonetes pugio | 5.6 | 4 | Other Nekton | 3.2 | 4 | Menidia menidia | 14.5 |
| 5 | Cyprinodon variegatus | 4.2 | 5 | Fundulus majalis | 2.3 | 5 | Callinectes sapidus | 5.7 |
| 6 | Fundulus majalis | 3.8 | 6 | Callinectes sapidus | 1.7 | 6 | Other Nekton | 3.1 |
| 7 | Menidia menidia | 2.5 | 7 | Palaemonetes pugio | 1.5 | 7 | Callinectes similis | 0.7 |
| 8 | Callinectes sapidus | 2.3 | 8 | Fundulus heteroclitus | 1.2 | 7 | Fundulus heteroclitus | 0.7 |
| 8 | Mugil cephalus | 2.3 | 9 | Menidia menidia | 0.6 | 7 | Paralichthys dentatus | 0.7 |
| 9 | Other Nekton | 2.1 | 10 | Mugil cephalus | 0.5 | 8 | Cyprinodon variegatus | 0.2 |
| 10 | Lucania parva | 1.8 | 10 | Palaemonetes intermedius | 0.5 | 8 | Mugil curema | 0.2 |
| 11 | Gobiosoma bosc | 1.1 | 11 | Paralichthys dentatus | 0.4 | 8 | Palaemonetes intermedius | 0.2 |
| 12 | Farfantepenaeus aztecus | 1.0 | 12 | Paralichthys albigutta | 0.1 | 8 | Paralichthys albigutta | 0.2 |
| 13 | Paralichthys dentatus | 0.8 | 13 | Callinectes similis | <0.1 | 9 | Palaemonetes pugio | 0.1 |
| 14 | Palaemonetes intermedius | 0.6 | 13 | Gobiosoma bosc | <0.1 | 10 | Eucinostomus spp. | 0.0 |
| 15 | Eucinostomus spp. | 0.5 | 13 | Lucania parva | <0.1 | 10 | Gobiosoma bosc | 0.0 |
| 16 | Mugil curema | 0.1 | 13 | Farfantepenaeus aztecus | <0.1 | 10 | Lucania parva | 0.0 |
| 17 | Paralichthys albigutta | <0.1 | 14 | Eucinostomus spp. | 0.0 | 10 | Mugil cephalus | 0.0 |
| 18 | Callinectes similis | 0.0 | 14 | Mugil curema | 0.0 | 10 | Farfantepenaeus aztecus | 0.0 |

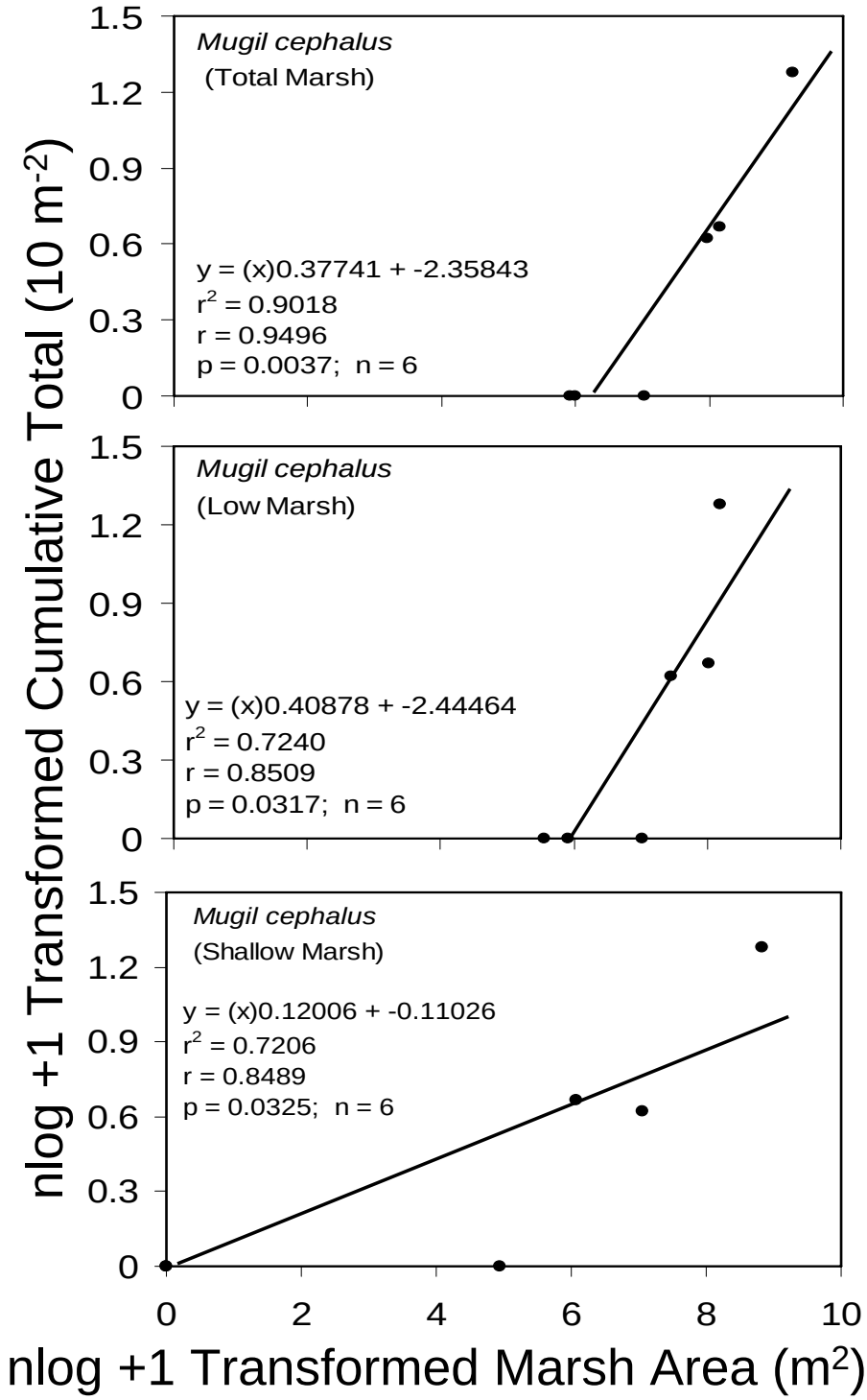
BIOMASS

| Mainland | | | Large Island | | | Small Island | | |
|---|---|---|---|---|---|---|---|---|
| RANK | SPECIES | % | RANK | SPECIES | % | RANK | SPECIES | % |
| 1 | Callinectes sapidus | 28.6 | 1 | Callinectes sapidus | 39.2 | 1 | Callinectes sapidus | 40.8 |
| 2 | Lagodon rhomboides | 19.9 | 2 | Lagodon rhomboides | 20.5 | 2 | Fundulus majalis | 37.5 |
| 3 | Leiostomus xanthurus | 10.4 | 3 | Leiostomus xanthurus | 14.0 | 3 | Lagodon rhomboides | 11.0 |
| 4 | Fundulus heteroclitus | 9.0 | 4 | Cyprinodon variegatus | 7.0 | 4 | Menidia menidia | 3.5 |
| 5 | Fundulus majalis | 8.5 | 5 | Fundulus majalis | 6.4 | 5 | Leiostomus xanthurus | 2.9 |
| 6 | Menidia menidia | 8.2 | 6 | Menidia menidia | 4.1 | 6 | Paralichthys dentatus | 2.4 |
| 7 | Cyprinodon variegatus | 4.3 | 7 | Fundulus heteroclitus | 3.1 | 7 | Paralichthys albigutta | 0.7 |
| 8 | Paralichthys dentatus | 2.6 | 8 | Paralichthys dentatus | 2.8 | 8 | Other Nekton | 0.6 |
| 9 | Lucania parva | 2.5 | 9 | Other Nekton | 1.4 | 9 | Fundulus heteroclitus | 0.3 |
| 10 | Farfantepenaeus aztecus | 1.9 | 10 | Paralichthys albigutta | 0.6 | 10 | Callinectes similis | 0.2 |
| 11 | Mugil cephalus | 1.4 | 11 | Mugil cephalus | 0.5 | 11 | Cyprinodon variegatus | 0.1 |
| 12 | Eucinostomus spp. | 0.9 | 12 | Palaemonetes pugio | 0.3 | 12 | Mugil curema | <0.1 |
| 13 | Paralichthys albigutta | 0.8 | 13 | Palaemonetes intermedius | 0.1 | 12 | Palaemonetes intermedius | <0.1 |
| 14 | Other Nekton | 0.4 | 14 | Callinectes similis | <0.1 | 12 | Palaemonetes pugio | <0.1 |
| 15 | Palaemonetes pugio | 0.2 | 14 | Gobiosoma bosc | <0.1 | 13 | Eucinostomus spp. | 0.0 |
| 15 | Gobiosoma bosc | 0.2 | 14 | Lucania parva | <0.1 | 13 | Gobiosoma bosc | 0.0 |
| 16 | Mugil curema | <0.1 | 14 | Farfantepenaeus aztecus | <0.1 | 13 | Lucania parva | 0.0 |
| 16 | Palaemonetes intermedius | <0.1 | 15 | Eucinostomus spp. | 0.0 | 13 | Mugil cephalus | 0.0 |
| 17 | Callinectes similis | 0.0 | 15 | Mugil curema | 0.0 | 13 | Farfantepenaeus aztecus | 0.0 |

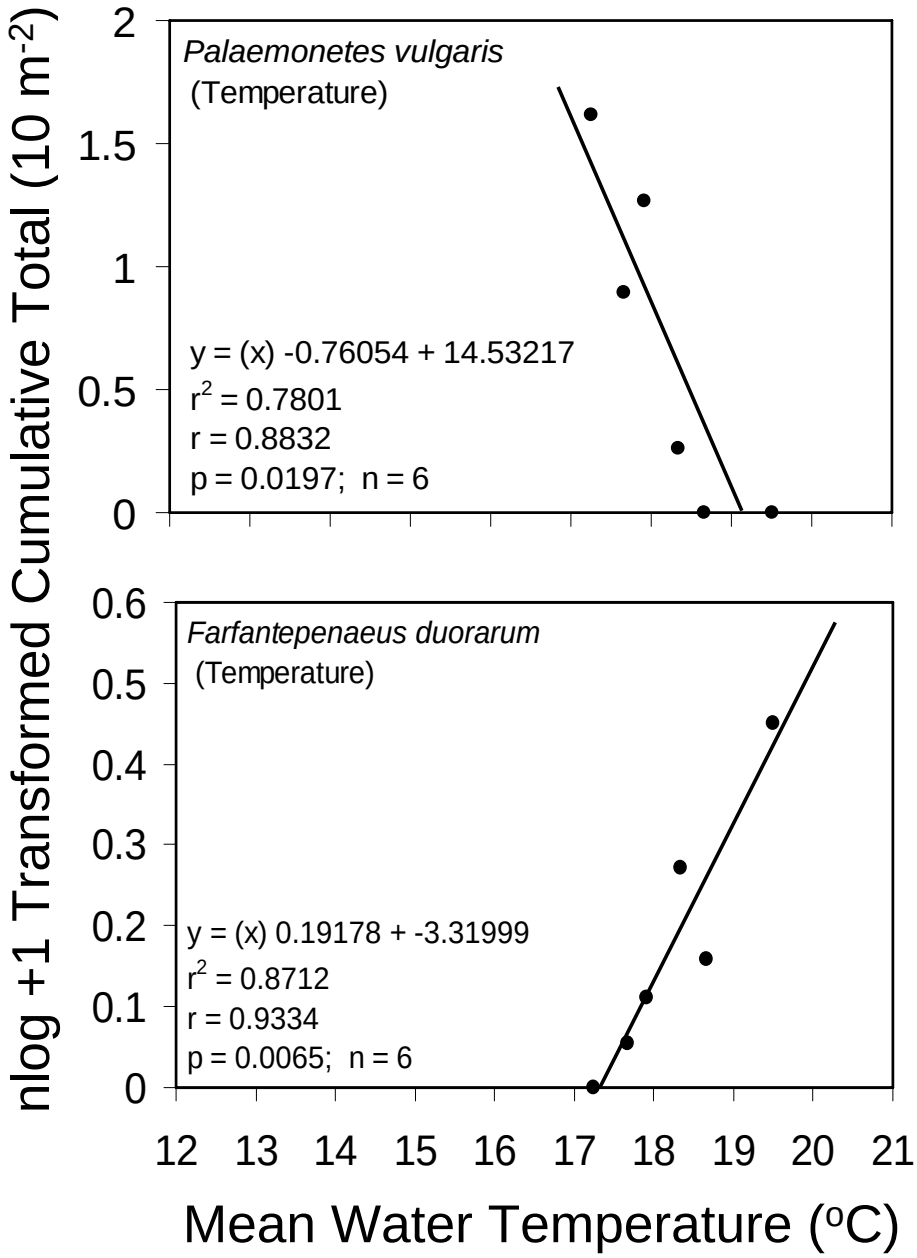
Appendix D. Principal component analysis for mean physical and biological parameters measured, and for cumulative nekton abundance for the all salt marsh types and for island salt marshes for year one and two combined. Eigenvector eigenvalues with the highest loading for individual attributes are indicated by an asterisk.

| Attribute | Principal Component 1 Main.& Islands / Islands | Principal Component 2 Main.& Islands / Islands | Principal Component 3 Main.& Islands / Islands | Principal Component 4 Main.& Islands / Islands |
|---|---|---|---|---|
| Nekton Abundance | -0.0788 / 0.0793 | 0.2544 / -0.2667 | -0.4421*/ -0.4441* | --- / 0.4478* |
| Salinity | -0.0011 / 0.0547 | 0.3052 / -0.0310 | 0.2134 / 0.5904* | --- / 0.4408* |
| Water Temperature | -0.2064 / -0.3503* | -0.4256*/ 0.0294 | 0.3090 / 0.1445 | --- / -0.3130 |
| Total Marsh | 0.4027* / 0.3564* | -0.0100 / 0.1366 | -0.0674 / 0.2336 | --- / 0.0362 |
| High Marsh Size | 0.3941* / 0.3252* | 0.0336 / 0.2818 | -0.1720 / -0.2193 | --- / 0.1607 |
| Low Marsh Size | 0.3946* / 0.3552* | 0.0243 / -0.0188 | -0.0381 / 0.2928 | --- / -0.0587 |
| Vertical Relief | -0.2040 / 0.3069* | 0.1773 / 0.0152 | -0.3618 / 0.3545 | --- / -0.0927 |
| Stem Density | 0.1236 / -0.2838 | -0.5085*/ 0.3296 | 0.1659 / -0.0399 | --- / 0.2180 |
| Canopy Height | 0.1629 / 0.2500 | 0.2530 / -0.1286 | 0.5578* / -0.1605 | --- / -0.5910* |
| Stem Diameter | 0.1873 / 0.3135* | 0.4811* / -0.3748 | 0.3263 / -0.1217 | --- / -0.0553 |
| Nearest Salt Marsh | -0.3655*/ 0.2815 | 0.1920 / -0.1944 | 0.0110 / -0.1677 | --- / 0.2394 |
| Silt-Clay Content | 0.3445* / 0.2793 | 0.0384 / 0.4204* | -0.1105 / -0.1189 | --- / -0.1018 |
| Organic Content | 0.3274* / 0.1276 | -0.1880 / 0.5901* | -0.1987 / -0.1866 | --- / 0.0222 |
| Eigenvalue | 5.5665 / 6.4177 | 2.4031 / 2.3783 | 1.5769 / 1.9062 | --- / 1.4615 |
| Proportion | 0.4282 / 0.4937 | 0.1849 / 0.1829 | 0.1213 / 0.1466 | --- / 0.1124 |

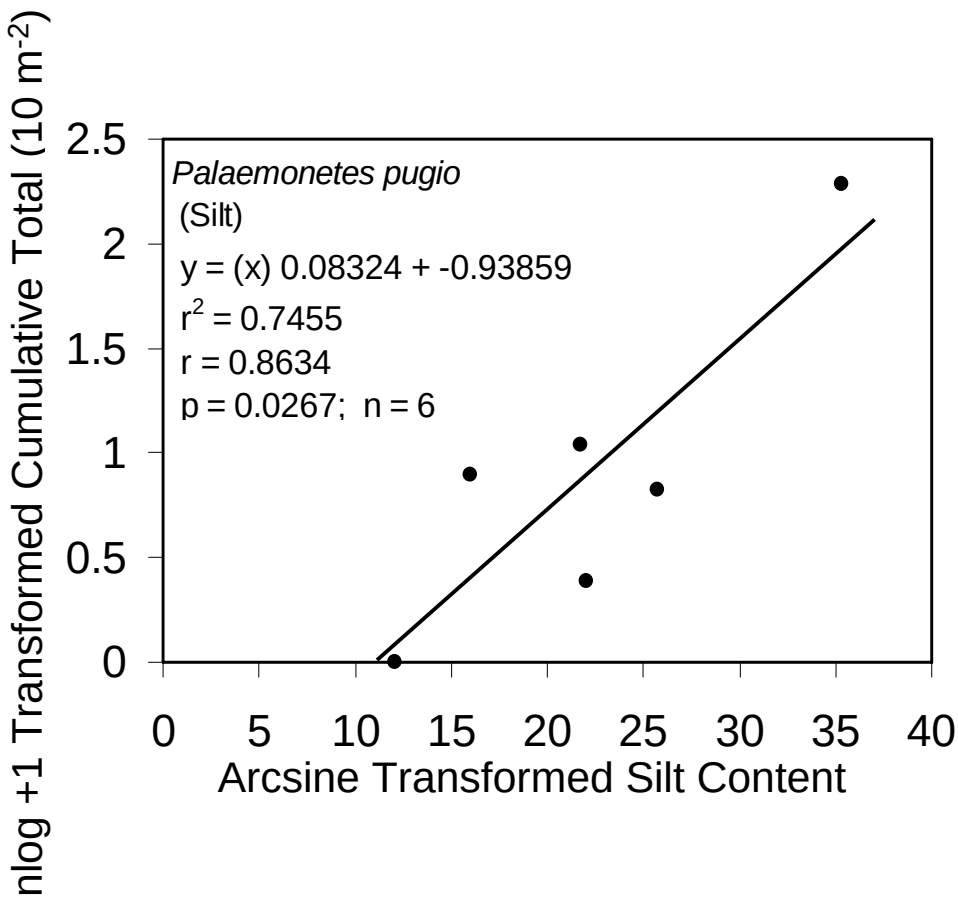
Appendix E. a-c. Linear regression analyses for nlog +1 transformed cumulative number of Mugil cephalus 10 $m^{-2}$ collected per island site compared to nlog +1 transformed total, low and shallow marsh areal size, $m^2$, per island. N = 6 and $p \leq 0.05$ for slope significance level.


Mugil cephalus
(Total Marsh)
y = (x)0.37741 + -2.35843
r2 = 0.9018
r = 0.9496
p = 0.0037; n = 6
Mugil cephalus
(Low Marsh)
y = (x)0.40878 + -2.44464
r2 = 0.7240
r = 0.8509
p = 0.0317; n = 6
Mugil cephalus
(Shallow Marsh)
y = (x)0.12006 + -0.11026
r2 = 0.7206
r = 0.8489
p = 0.0325; n = 6
nlog +1 Transformed Cumulative Total (10 m-2)
nlog +1 Transformed Marsh Area (m2)
0
0.3
0.6
0.9
1.2
1.5
0
2
4
6
8
10

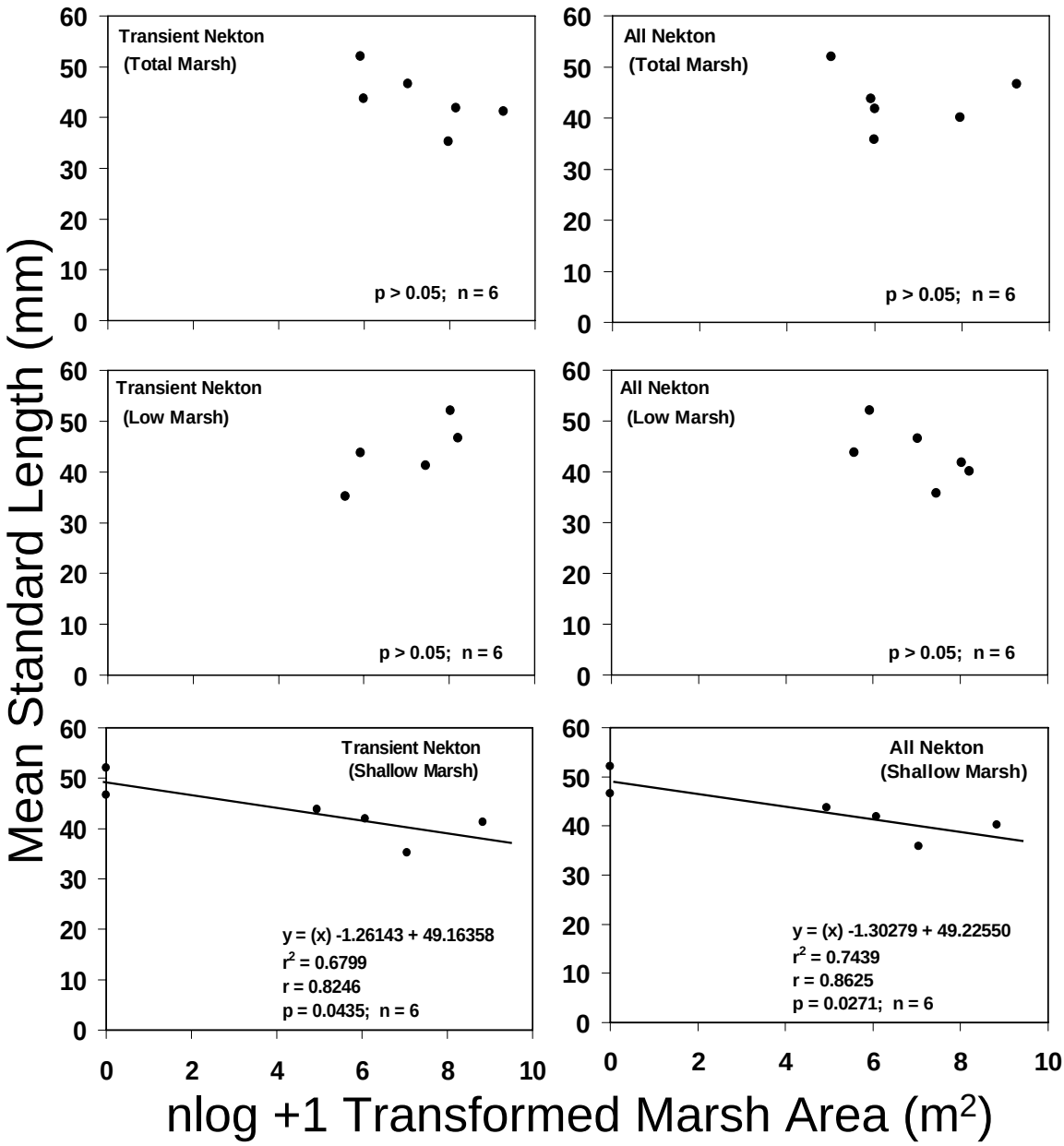
Appendix F. a,b. Linear regression analyses for nlog +1 transformed cumulative number of Palaemonetes vulgaris and Farfantepenaeus duorarum 10 $m^{-2}$ collected per island site compared to mean water temperature (°C) per island. N = 6 and $p \leq 0.05$ for slope significance level.


nlog +1 Transformed Cumulative Total (10 m-2)
Palaemonetes vulgaris
(Temperature)
y = (x) -0.76054 + 14.53217
r2 = 0.7801
r = 0.8832
p = 0.0197; n = 6
Farfantepenaeus duorarum
(Temperature)
y = (x) 0.19178 + -3.31999
r2 = 0.8712
r = 0.9334
p = 0.0065; n = 6
0
0.5
1
1.5
2
0
0.1
0.2
0.3
0.4
0.5
0.6
12
13
14
15
16
17
18
19
20
21
Mean Water Temperature (oC)

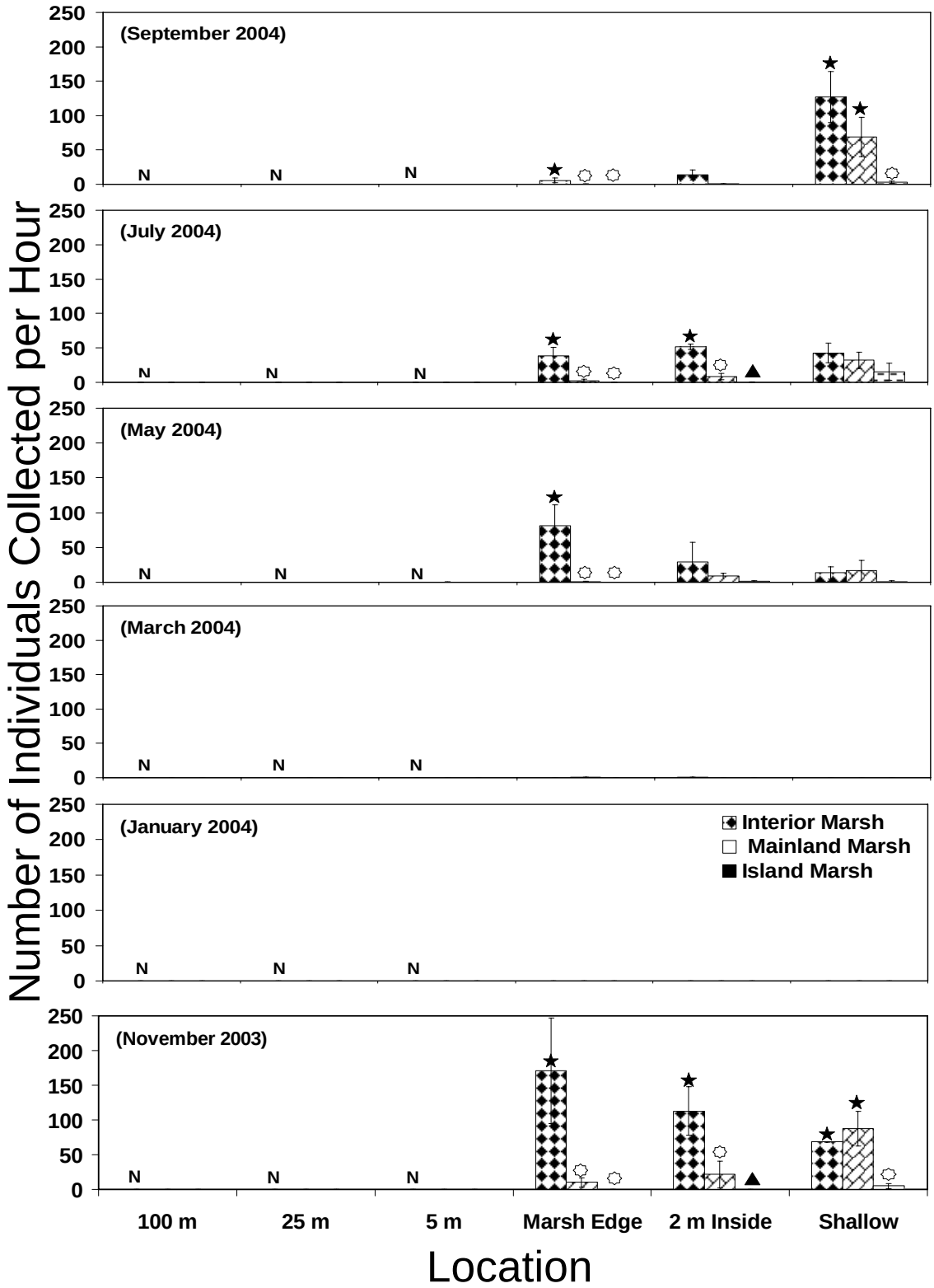
Appendix G. Linear regression analyses for nlog +1 transformed cumulative number of Palaemonetes pugio 10 $m^{-2}$ collected per island site compared to arcsine transformed sediment silt-clay content per island. N = 6 and $p \leq 0.05$ for slope significance level.


nlog +1 Transformed Cumulative Total (10 m-2)
2.5
2
1.5
1
0.5
0
Palaemonetes pugio
(Silt)
y = (x) 0.08324 + -0.93859
r2 = 0.7455
r = 0.8634
p = 0.0267; n = 6
0
5
10
15
20
25
30
35
40
Arcsine Transformed Silt Content

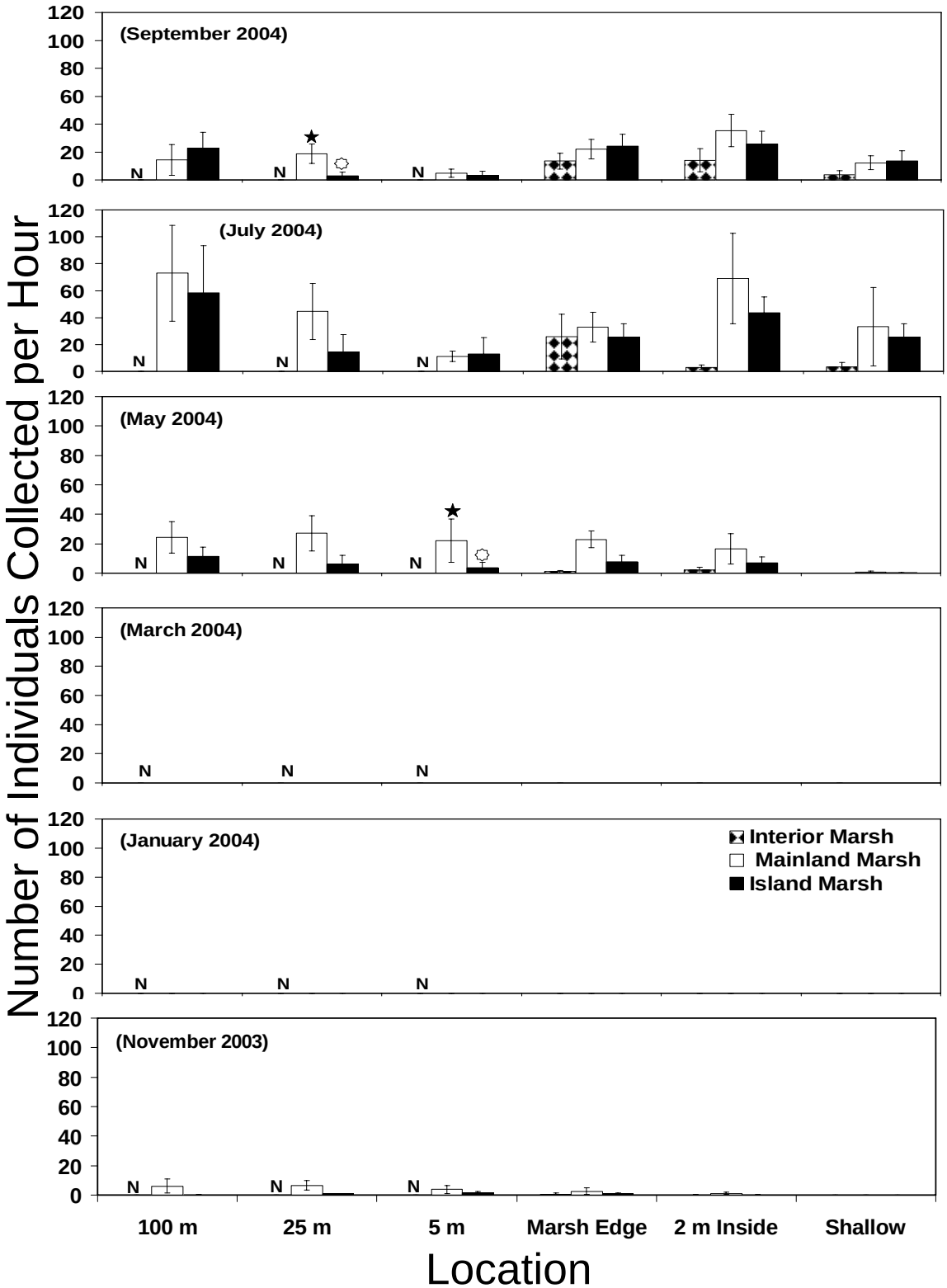
Appendix H. a-f. Linear regression analyses for mean standard length (mm) of transient nekton and all nekton species combined per island site compared to nlog +1 transformed total, low and shallow marsh areal size, $m^2$, per island. N = 6 and $p \leq 0.05$ for slope significance level.


Transient Nekton
(Total Marsh)
p > 0.05; n = 6
All Nekton
(Total Marsh)
p > 0.05; n = 6
Transient Nekton
(Low Marsh)
p > 0.05; n = 6
All Nekton
(Low Marsh)
p > 0.05; n = 6
Transient Nekton
(Shallow Marsh)
y = (x) -1.26143 + 49.16358
r2 = 0.6799
r = 0.8246
p = 0.0435; n = 6
All Nekton
(Shallow Marsh)
y = (x) -1.30279 + 49.22550
r2 = 0.7439
r = 0.8625
p = 0.0271; n = 6
Mean Standard Length (mm)
nlog +1 Transformed Marsh Area (m2)

Appendix I. Comparison between salt marsh types for mean number of Fundulus heteroclitus collected per hour with eel pots based on location and date. Location comparisons between salt marsh types, for each date, that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. Eel pots were set at 100, 25 and 5 m in shallow water flats seaward of the salt marsh, the salt marsh edge, 2 meters inside and in the shallow marsh areas (shallow) at each mainland and island site, and in the marsh creek, the salt marsh edge, 2 meters inside and in the shallow marsh areas (shallow) at each interior salt marsh site. N indicates that no samples were collected for that salt marsh type. One standard error is indicated by the error bars.


(September 2004)
(July 2004)
(May 2004)
(March 2004)
(January 2004)
(November 2003)
Interior Marsh
Mainland Marsh
Island Marsh
N
250
200
150
100
50
0
Number of Individuals Collected per Hour
100 m
25 m
5 m
Marsh Edge
2 m Inside
Shallow
Location

Appendix J. Comparison between salt marsh types for mean number of Lagodon rhomboides collected per hour with eel pots based on location and date. Location comparisons between salt marsh types, for each date, that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. Eel pots were set at 100, 25 and 5 m in shallow water flats seaward of the salt marsh, the salt marsh edge, 2 meters inside and in the shallow marsh areas (shallow) at each mainland and island site, and in the marsh creek, the salt marsh edge, 2 meters inside and in the shallow marsh areas (shallow) at each interior salt marsh site. N indicates that no samples were collected for that salt marsh type. One standard error is indicated by the error bars.


(September 2004)
(July 2004)
(May 2004)
(March 2004)
(January 2004)
(November 2003)
Interior Marsh
Mainland Marsh
Island Marsh
Number of Individuals Collected per Hour
0
20
40
60
80
100
120
N
100 m
25 m
5 m
Marsh Edge
2 m Inside
Shallow
Location

Appendix K. Linear regression results (natural log transformed data) examining average water depth versus catch per hour (CPH) for Fundulus heteroclitus and Lagodon rhomboides from high tide eel pot collections for the various salt marsh types sampled. NS = no significant difference at the $p \leq 0.05$ level.

| | | Water Depth (all traps) | | | | | | Water Depth (marsh traps only) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Date | r | Intercept | slope | $r^2$ | p | n | r | Intercept | slope | $r^2$ | p | n |
| | | Interior Marshes | | | | | | | | | | | |
| F. heteroclitus CPH | 11/03 | NS | NS | NS | NS | NS | 11 | NS | NS | NS | NS | NS | 8 |
| F. heteroclitus CPH | 1/04 | NS | NS | NS | NS | NS | 12 | NS | NS | NS | NS | NS | 9 |
| F. heteroclitus CPH | 3/04 | NS | NS | NS | NS | NS | 10 | NS | NS | NS | NS | NS | 7 |
| F. heteroclitus CPH | 5/04 | NS | NS | NS | NS | NS | 10 | 0.92766 | -0.23247 | 0.19924 | 0.8606 | 0.0026 | 7 |
| F. heteroclitus CPH | 7/04 | -0.68144 | 5.02151 | -0.05290 | 0.4644 | 0.0147 | 12 | NS | NS | NS | NS | NS | 9 |
| F. heteroclitus CPH | 9/04 | -0.85501 | 6.05155 | -0.09651 | 0.7310 | 0.0004 | 12 | -0.80934 | 7.16235 | -0.13374 | 0.6550 | 0.0082 | 9 |
| L. rhomboides CPH | 11/03 | 0.62899 | -0.38694 | 0.03659 | 0.3956 | 0.0382 | 11 | NS | NS | NS | NS | NS | 8 |
| L. rhomboides CPH | 1/04 | NS | NS | NS | NS | NS | 12 | NS | NS | NS | NS | NS | 9 |
| L. rhomboides CPH | 3/04 | NS | NS | NS | NS | NS | 10 | NS | NS | NS | NS | NS | 7 |
| L. rhomboides CPH | 5/04 | NS | NS | NS | NS | NS | 10 | NS | NS | NS | NS | NS | 7 |
| L. rhomboides CPH | 7/04 | NS | NS | NS | NS | NS | 12 | NS | NS | NS | NS | NS | 9 |
| L. rhomboides CPH | 9/04 | NS | NS | NS | NS | NS | 12 | NS | NS | NS | NS | NS | 9 |

Appendix K. (Continued).

| Species | Date | Water Depth (all traps) r | Intercept | slope | $r^2$ | p | n | Water Depth (marsh traps only) r | Intercept | slope | $r^2$ | p | n |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | Mainland Marshes | | | | | | | |
| F. heteroclitus CPH | 11/03 | -0.56732 | 2.30773 | -0.02428 | 0.3219 | <0.0001 | 42 | -0.75341 | 4.82015 | -0.10672 | 0.5676 | 0.0003 | 18 |
| F. heteroclitus CPH | 1/04 | NS | NS | NS | NS | NS | 42 | NS | NS | NS | NS | NS | 18 |
| F. heteroclitus CPH | 3/04 | NS | NS | NS | NS | NS | 36 | -0.57984 | 0.70951 | -0.01874 | 0.3362 | 0.0481 | 12 |
| F. heteroclitus CPH | 5/04 | -0.46353 | 1.20171 | -0.01040 | 0.2149 | 0.0026 | 40 | NS | NS | NS | NS | NS | 16 |
| F. heteroclitus CPH | 7/04 | -0.52405 | 1.78602 | -0.01510 | 0.2746 | 0.0004 | 41 | -0.75322 | 5.09432 | -0.08867 | 0.5673 | 0.0005 | 17 |
| F. heteroclitus CPH | 9/04 | -0.49195 | 1.77281 | -0.01382 | 0.2420 | 0.0009 | 42 | -0.76250 | 5.28387 | -0.08277 | 0.5814 | 0.0002 | 18 |
| L. rhomboides CPH | 11/03 | NS | NS | NS | NS | NS | 42 | NS | NS | NS | NS | NS | 18 |
| L. rhomboides CPH | 1/04 | NS | NS | NS | NS | NS | 42 | NS | NS | NS | NS | NS | 18 |
| L. rhomboides CPH | 3/04 | NS | NS | NS | NS | NS | 36 | NS | NS | NS | NS | NS | 12 |
| L. rhomboides CPH | 5/04 | NS | NS | NS | NS | NS | 40 | 0.79262 | -1.08954 | 0.10384 | 0.6282 | 0.0003 | 16 |
| L. rhomboides CPH | 7/04 | -0.41118 | 3.59515 | -0.01659 | 0.1691 | 0.0076 | 41 | NS | NS | NS | NS | NS | 17 |
| L. rhomboides CPH | 9/04 | -0.41363 | 2.96231 | -0.01325 | 0.1711 | 0.0065 | 42 | NS | NS | NS | NS | NS | 18 |

Appendix K. Continued.

| Species | Date | Water Depth (all traps) r | Intercept | slope | $r^2$ | p | n | Water Depth (marsh traps only) r | Intercept | slope | $r^2$ | p | n |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Island Marshes | | | | | | | | | | | |
| F. heteroclitus CPH | 11/03 | NS | NS | NS | NS | NS | 41 | NS | NS | NS | NS | NS | 18 |
| F. heteroclitus CPH | 1/04 | NS | NS | NS | NS | NS | 41 | NS | NS | NS | NS | NS | 18 |
| F. heteroclitus CPH | 3/04 | NS | NS | NS | NS | NS | 36 | NS | NS | NS | NS | NS | 13 |
| F. heteroclitus CPH | 5/04 | -0.30918 | 0.33306 | -0.00321 | 0.0956 | 0.05 | 40 | NS | NS | NS | NS | NS | 17 |
| F. heteroclitus CPH | 7/04 | -0.32158 | 0.6476 | -0.00562 | 0.1034 | 0.043 | 40 | NS | NS | NS | NS | NS | 17 |
| F. heteroclitus CPH | 9/04 | -0.32862 | 0.15481 | -0.00346 | 0.1080 | 0.0359 | 41 | -0.50160 | 1.25506 | -0.01917 | 0.2516 | 0.03339 | 18 |
| L. rhomboides CPH | 11/03 | NS | NS | NS | NS | NS | 41 | 0.47968 | -0.08572 | 0.01435 | 0.2227 | 0.048 | 18 |
| L. rhomboides CPH | 1/04 | NS | NS | NS | NS | NS | 41 | NS | NS | NS | NS | NS | 18 |
| L. rhomboides CPH | 3/04 | NS | NS | NS | NS | NS | 36 | NS | NS | NS | NS | NS | 13 |
| L. rhomboides CPH | 5/04 | NS | NS | NS | NS | NS | 40 | NS | NS | NS | NS | NS | 17 |
| L. rhomboides CPH | 7/04 | -0.45753 | 3.36223 | -0.01811 | 0.2093 | 0.003 | 40 | NS | NS | NS | NS | NS | 17 |
| L. rhomboides CPH | 9/04 | -0.35865 | 2.49269 | -0.01172 | 0.1286 | 0.0213 | 41 | NS | NS | NS | NS | NS | 18 |

Appendix K. Continued.

| Species | Date | Water Depth (all traps) r | Intercept | slope | $r^2$ | p | n | Water Depth (marsh traps only) r | Intercept | slope | $r^2$ | p | n |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | All Marshes | | | | | | | | | | | |
| F. heteroclitus CPH | 11/03 | -0.46549 | 2.10211 | -0.02093 | 0.2167 | <0.0001 | 94 | -0.48387 | 3.63292 | -0.07551 | 0.2341 | 0.0009 | 44 |
| F. heteroclitus CPH | 1/04 | NS | NS | NS | NS | NS | 95 | NS | NS | NS | NS | NS | 45 |
| F. heteroclitus CPH | 3/04 | NS | NS | NS | NS | NS | 82 | -0.35147 | 0.31044 | -0.00689 | 0.1235 | 0.0485 | 32 |
| F. heteroclitus CPH | 5/04 | -0.41219 | 1.37737 | -0.01203 | 0.1699 | <0.0001 | 90 | -0.39268 | 2.32674 | -0.03988 | 0.1542 | 0.0122 | 40 |
| F. heteroclitus CPH | 7/04 | -0.50341 | 1.94265 | -0.01651 | 0.2534 | <0.0001 | 93 | -0.64710 | 4.42249 | -0.07400 | 0.4187 | <0.0001 | 43 |
| F. heteroclitus CPH | 9/04 | -0.46071 | 1.59675 | -0.01278 | 0.2123 | <0.0001 | 95 | -0.64795 | 4.04133 | -0.06260 | 0.4198 | <0.0001 | 45 |
| L. rhomboides CPH | 11/03 | NS | NS | NS | NS | NS | 94 | 0.41155 | -0.10548 | 0.01818 | 0.1694 | 0.0055 | 44 |
| L. rhomboides CPH | 1/04 | NS | NS | NS | NS | NS | 95 | NS | NS | NS | NS | NS | 45 |
| L. rhomboides CPH | 3/04 | NS | NS | NS | NS | NS | 82 | NS | NS | NS | NS | NS | 32 |
| L. rhomboides CPH | 5/04 | NS | NS | NS | NS | NS | 90 | 0.48887 | -0.10621 | 0.04628 | 0.2390 | 0.0014 | 40 |
| L. rhomboides CPH | 7/04 | -0.34220 | 3.06631 | -0.01365 | 0.1171 | 0.0008 | 93 | NS | NS | NS | NS | NS | 43 |
| L. rhomboides CPH | 9/04 | -0.35152 | 2.61840 | -0.01132 | 0.1236 | 0.0005 | 95 | NS | NS | NS | NS | NS | 45 |

Appendix L. Linear regression results examining average water depth versus mean fish size per eel pot for Fundulus heteroclitus and Lagodon rhomboides from high tide eel pot collections for the various salt marsh types sampled. NS = no significant difference at the $p \leq 0.05$ level.

| Species | Date | Water Depth (all traps) r | Intercept | slope | $r^2$ | p | n | Water Depth (marsh traps only) r | Intercept | slope | $r^2$ | p | n |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Interior Marshes | | | | | | | | | | | |
| F. heteroclitus | 11/03 | 0.78033 | 36.88927 | 0.33547 | 0.6089 | 0.0046 | 11 | NS | NS | NS | NS | NS | 8 |
| F. heteroclitus | 1/04 | NS | NS | NS | NS | NS | 1 | NS | NS | NS | NS | NS | 1 |
| F. heteroclitus | 3/04 | NS | NS | NS | NS | NS | 2 | NS | NS | NS | NS | NS | 1 |
| F. heteroclitus | 5/04 | 0.67789 | 42.57284 | 0.25039 | 0.4595 | 0.0448 | 9 | NS | NS | NS | NS | NS | 6 |
| F. heteroclitus | 7/04 | 0.77359 | 34.44190 | 0.44825 | 0.5984 | 0.0052 | 11 | 0.69029 | 28.99694 | 0.66890 | 0.4765 | 0.0396 | 9 |
| F. heteroclitus | 9/04 | NS | NS | NS | NS | NS | 7 | NS | NS | NS | NS | NS | 7 |
| F. heteroclitus | ALL | 0.61984 | 35.50898 | 0.36917 | 0.3842 | <0.0001 | 41 | 0.37323 | 37.30265 | 0.28007 | 0.1393 | 0.0354 | 32 |
| L. rhomboides | 11/03 | NS | NS | NS | NS | NS | 4 | NS | NS | NS | NS | NS | 2 |
| L. rhomboides | 1/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 3/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 5/04 | NS | NS | NS | NS | NS | 5 | NS | NS | NS | NS | NS | 3 |
| L. rhomboides | 7/04 | 0.70874 | 33.53850 | 0.27968 | 0.5023 | 0.0491 | 8 | NS | NS | NS | NS | NS | 5 |
| L. rhomboides | 9/04 | NS | NS | NS | NS | NS | 10 | NS | NS | NS | NS | NS | 7 |
| L. rhomboides | ALL | 0.39698 | 35.94759 | 0.30536 | 0.1576 | 0.0403 | 27 | 0.48695 | 31.20809 | 0.55643 | 0.2371 | 0.0474 | 17 |

Appendix L. (Continued).

| Species | Date | Water Depth (all traps) r | Intercept | slope | $r^2$ | p | n | Water Depth (marsh traps only) r | Intercept | slope | $r^2$ | p | n |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | Mainland Marshes | | | | | | | |
| F. heteroclitus | 11/03 | NS | NS | NS | NS | NS | 12 | NS | NS | NS | NS | NS | 12 |
| F. heteroclitus | 1/04 | NS | NS | NS | NS | NS | 1 | NS | NS | NS | NS | NS | 0 |
| F. heteroclitus | 3/04 | NS | NS | NS | NS | NS | 2 | NS | NS | NS | NS | NS | 2 |
| F. heteroclitus | 5/04 | NS | NS | NS | NS | NS | 12 | NS | NS | NS | NS | NS | 11 |
| F. heteroclitus | 7/04 | 0.73732 | 37.40399 | 0.44470 | 0.5436 | 0.0062 | 12 | 0.73732 | 37.40399 | 0.44470 | 0.5436 | 0.0062 | 12 |
| F. heteroclitus | 9/04 | 0.75643 | 23.36947 | 0.54875 | 0.5795 | 0.0105 | 10 | 0.75643 | 23.36947 | 0.54875 | 0.5795 | 0.0105 | 10 |
| F. heteroclitus | ALL | 0.58929 | 38.94717 | 0.33311 | 0.3473 | <0.0001 | 49 | 0.55931 | 39.07283 | 0.32134 | 0.3128 | <0.0001 | 47 |
| L. rhomboides | 11/03 | NS | NS | NS | NS | NS | 21 | NS | NS | NS | NS | NS | 5 |
| L. rhomboides | 1/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 3/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 5/04 | NS | NS | NS | NS | NS | 29 | NS | NS | NS | NS | NS | 10 |
| L. rhomboides | 7/04 | 0.60792 | 37.33138 | 0.18658 | 0.3692 | 0.0002 | 33 | NS | NS | NS | NS | NS | 16 |
| L. rhomboides | 9/04 | 0.67444 | 44.21658 | 0.23843 | 0.4526 | 0.0001 | 27 | 0.56313 | 41.62840 | 0.29757 | 0.3171 | 0.0288 | 15 |
| L. rhomboides | ALL | 0.28103 | 39.55079 | 0.12613 | 0.0790 | 0.0029 | 110 | 0.50647 | 29.23889 | 0.41537 | 0.2565 | 0.0003 | 46 |

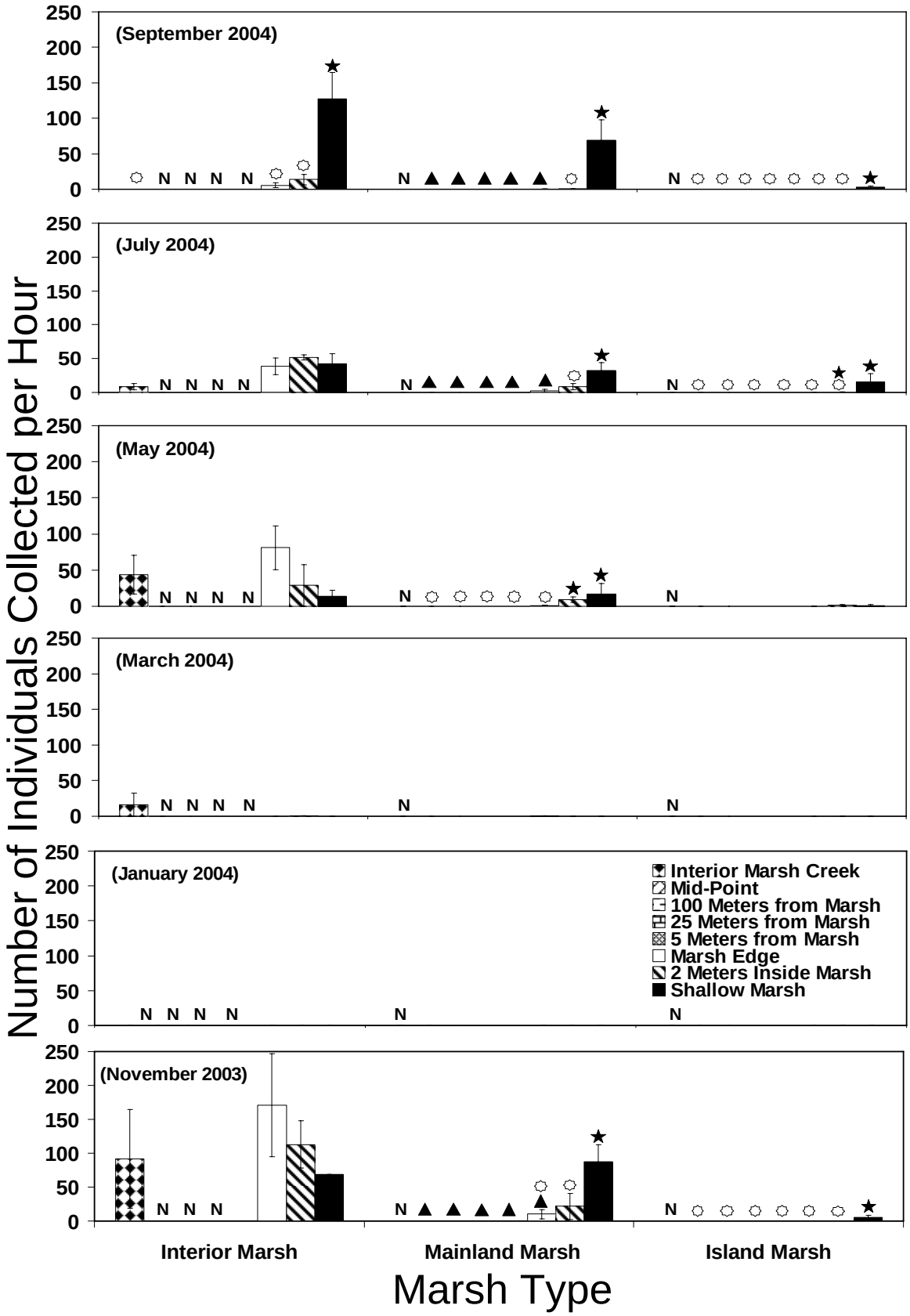
Appendix L. Continued.

| Species | Date | Water Depth (all traps) r | Intercept | slope | $r^2$ | p | n | Water Depth (marsh traps only) r | Intercept | slope | $r^2$ | p | n |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Island Marshes | | | | | | | | | | | |
| F. heteroclitus | 11/03 | NS | NS | NS | NS | NS | 3 | NS | NS | NS | NS | NS | 3 |
| F. heteroclitus | 1/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| F. heteroclitus | 3/04 | NS | NS | NS | NS | NS | 1 | NS | NS | NS | NS | NS | 1 |
| F. heteroclitus | 5/04 | 0.99996 | 29.62712 | 1.20869 | 0.9999 | 0.0059 | 3 | 0.99996 | 29.62712 | 1.20869 | 0.9999 | 0.0059 | 3 |
| F. heteroclitus | 7/04 | NS | NS | NS | NS | NS | 6 | NS | NS | NS | NS | NS | 6 |
| F. heteroclitus | 9/04 | NS | NS | NS | NS | NS | 3 | NS | NS | NS | NS | NS | 3 |
| F. heteroclitus | ALL | NS | NS | NS | NS | NS | 16 | NS | NS | NS | NS | NS | 16 |
| L. rhomboides | 11/03 | NS | NS | NS | NS | NS | 16 | NS | NS | NS | NS | NS | 6 |
| L. rhomboides | 1/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 3/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 5/04 | NS | NS | NS | NS | NS | 19 | NS | NS | NS | NS | NS | 9 |
| L. rhomboides | 7/04 | 0.58674 | 41.68879 | 0.15545 | 0.3443 | 0.0010 | 28 | NS | NS | NS | NS | NS | 16 |
| L. rhomboides | 9/04 | 0.51015 | 54.03850 | 0.14118 | 0.2603 | 0.0129 | 23 | NS | NS | NS | NS | NS | 15 |
| L. rhomboides | ALL | NS | NS | NS | NS | NS | 86 | NS | NS | NS | NS | NS | 46 |

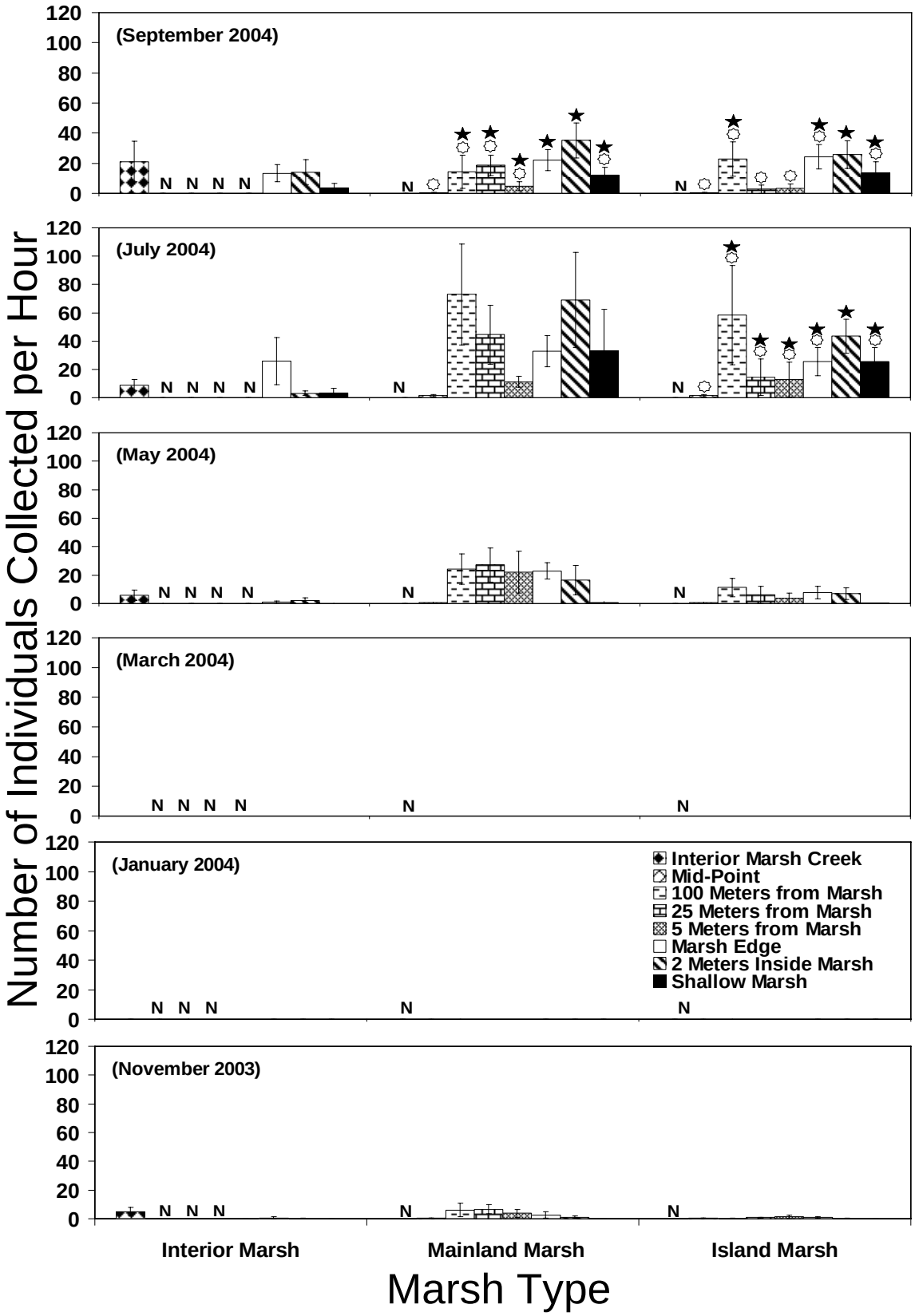
Appendix L. (Continued).

| Species | Date | Water Depth (all traps) r | Intercept | slope | $r^2$ | p | n | Water Depth (marsh traps only) r | Intercept | slope | $r^2$ | p | n |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | All Marshes | | | | | | | | | | | |
| F. heteroclitus | 11/03 | 0.40614 | 40.42094 | 0.23151 | 0.1650 | 0.0395 | 26 | NS | NS | NS | NS | NS | 23 |
| F. heteroclitus | 1/04 | NS | NS | NS | NS | NS | 2 | NS | NS | NS | NS | NS | 45 |
| F. heteroclitus | 3/04 | NS | NS | NS | NS | NS | 5 | NS | NS | NS | NS | NS | 5 |
| F. heteroclitus | 5/04 | 0.53551 | 45.61409 | 0.23278 | 0.2868 | 0.007 | 24 | 0.51528 | 45.24851 | 0.24401 | 0.2655 | 0.0201 | 20 |
| F. heteroclitus | 7/04 | 0.37859 | 43.15067 | 0.27476 | 0.1433 | 0.0428 | 29 | NS | NS | NS | NS | NS | 27 |
| F. heteroclitus | 9/04 | 0.47354 | 33.13929 | 0.34751 | 0.2242 | 0.0349 | 20 | 0.47354 | 33.13929 | 0.34751 | 0.2242 | 0.0349 | 20 |
| F. heteroclitus | ALL | 0.46042 | 40.15834 | 0.29433 | 0.2120 | <0.0001 | 106 | 0.39843 | 40.47856 | 0.27895 | 0.1587 | <0.0001 | 95 |
| L. rhomboides | 11/03 | NS | NS | NS | NS | NS | 41 | NS | NS | NS | NS | NS | 13 |
| L. rhomboides | 1/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 3/04 | NS | NS | NS | NS | NS | 0 | NS | NS | NS | NS | NS | 0 |
| L. rhomboides | 5/04 | NS | NS | NS | NS | NS | 53 | NS | NS | NS | NS | NS | 22 |
| L. rhomboides | 7/04 | 0.60957 | 38.87109 | 0.17730 | 0.3716 | <0.0001 | 69 | 0.50372 | 37.08714 | 0.22874 | 0.2537 | 0.0015 | 37 |
| L. rhomboides | 9/04 | 0.57471 | 49.30733 | 0.18669 | 0.3303 | <0.0001 | 60 | 0.38687 | 48.53332 | 0.20002 | 0.1497 | 0.018 | 37 |
| L. rhomboides | ALL | 0.21469 | 43.14517 | 0.09424 | 0.0461 | 0.0013 | 223 | 0.33648 | 36.98897 | 0.26391 | 0.1132 | 0.0003 | 109 |

Appendix M. Mean number of Fundulus heteroclitus collected per hour via eel pots based on salt marsh and shallow water flat location, by salt marsh type and date. Location mean comparisons within a salt marsh type, for each date, that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. N indicates that no samples were collected for that salt marsh type. One standard error is indicated by the error bars.


(September 2004)
(July 2004)
(May 2004)
(March 2004)
(January 2004)
(November 2003)
250
200
150
100
50
0
N
Interior Marsh Creek
Mid-Point
100 Meters from Marsh
25 Meters from Marsh
5 Meters from Marsh
Marsh Edge
2 Meters Inside Marsh
Shallow Marsh
Number of Individuals Collected per Hour
Interior Marsh
Mainland Marsh
Island Marsh
Marsh Type

Appendix N. Mean number of Lagodon rhomboides collected per hour via eel pots based on salt marsh and shallow water flat location, by salt marsh type and date. Location mean comparisons within a salt marsh type, for each date, that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. N indicates that no samples were collected for that salt marsh type. One standard error is indicated by the error bars.


(September 2004)
(July 2004)
(May 2004)
(March 2004)
(January 2004)
(November 2003)
Number of Individuals Collected per Hour
0
20
40
60
80
100
120
N
Interior Marsh Creek
Mid-Point
100 Meters from Marsh
25 Meters from Marsh
5 Meters from Marsh
Marsh Edge
2 Meters Inside Marsh
Shallow Marsh
Interior Marsh
Mainland Marsh
Island Marsh
Marsh Type

Appendix O. Mean frequency of occurrence for Fundulus heteroclitus and Lagodon rhomboides at set positions from salt marsh during high and ebb tide eel pot collections. For each species comparisons between high (H) and ebb (E) tide stage for each marsh type mainland salt marsh (mainland), island salt marsh (island) and interior salt marsh (interior) that are significantly different from one another are indicated by asterisks ($p \leq 0.05$).

| | | Location | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | 25 m | | | | 100 m | | | | Mid-Point | | | | Marsh Creek | |
| | | Mainland | | Island | | Mainland | | Island | | Mainland | | Island | | Interior | |
| Species | Date | H | L | H | L | H | L | H | L | H | L | H | L | H | L |
| Fundulus heteroclitus | 11/03 | 0.0* | 0.7* | 0.0 | 0.3 | 0.0 | 0.2 | 0.0 | 0.2 | 0.0 | 0.0 | 0.0 | 0.0 | 1.0 | 1.0 |
| Fundulus heteroclitus | 1/04 | 0.2 | 0.2 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.7 |
| Fundulus heteroclitus | 3/04 | 0.0* | 0.7* | 0.0 | 0.2 | 0.0 | 0.2 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.3 | 0.7 |
| Fundulus heteroclitus | 5//04 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 1.0 | 1.0 |
| Fundulus heteroclitus | 7/04 | 0.0 | 0.3 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.7 | 1.0 |
| Fundulus heteroclitus | 9/04 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.7 |
| Lagodon rhomboides | 11/03 | 0.8 | 0.8 | 0.7 | 0.5 | 0.8 | 0.7 | 0.4 | 0.8 | 0.3 | 0.5 | 0.3 | 0.5 | 0.7 | 0.0 |
| Lagodon rhomboides | 1/04 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Lagodon rhomboides | 3/04 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Lagodon rhomboides | 5/04 | 0.8 | 1.0 | 0.3 | 0.3 | 0.8 | 0.8 | 0.6 | 0.6 | 0.7 | 0.5 | 0.7 | 0.5 | 0.7 | 0.3 |
| Lagodon rhomboides | 7/04 | 0.7 | 1.0 | 0.5 | 0.8 | 0.7 | 1.0 | 0.8 | 1.0 | 0.5 | 0.5 | 0.5 | 0.5 | 1.0 | 1.0 |
| Lagodon rhomboides | 9/04 | 0.8 | 1.0 | 0.2* | 0.8* | 0.5 | 0.8 | 0.8 | 1.0 | 0.2* | 0.8* | 0.2* | 0.8* | 1.0 | 1.0 |

Appendix P. Linear regression analysis, per sampling period, examining catch per hour (CPH) versus average water to estimate water depth intercepts (critical water depths) for zero CPH for Fundulus heteroclitus and Lagodon rhomboides from high tide eel pot collections for the various marsh types sampled. Only intercepts for significant regressions ($p \leq 0.05$) are shown. Maximum depths are represented by ($\leq$) adjacent to them while minimum depth are represented by a ($\geq$) adjacent to them. NS = no significant difference at the $p = 0.05$ level.

| Species | Date | Water Depth (all traps) Intercept | p | n | Water Depth (marsh traps only) Intercept | p | n |
|---|---|---|---|---|---|---|---|
| | | | All Marshes | | | | |
| Fundulus heteroclitus CPH | 11/03 | $\leq$ 56.6 | <0.0001 | 94 | $\leq$ 24.8 | 0.0009 | 44 |
| Fundulus heteroclitus CPH | 1/04 | NS | NS | 95 | NS | NS | 45 |
| Fundulus heteroclitus CPH | 3/04 | NS | NS | 82 | $\leq$ 29.8 | 0.0485 | 32 |
| Fundulus heteroclitus CPH | 5/04 | $\leq$ 64.0 | <0.0001 | 90 | $\leq$ 30.0 | 0.0122 | 40 |
| Fundulus heteroclitus CPH | 7/04 | $\leq$ 75.8 | <0.0001 | 93 | $\leq$ 43.2 | <0.0001 | 43 |
| Fundulus heteroclitus CPH | 9/04 | $\leq$ 83.1 | <0.0001 | 95 | $\leq$ 48.2 | <0.0001 | 45 |
| Lagodon rhomboides CPH | 11/03 | NS | NS | 94 | $\geq$ 21.0 | 0.0055 | 44 |
| Lagodon rhomboides CPH | 1/04 | NS | NS | 95 | NS | NS | 45 |
| Lagodon rhomboides CPH | 3/04 | NS | NS | 82 | NS | NS | 32 |
| Lagodon rhomboides CPH | 5/04 | NS | NS | 90 | $\geq$ 24.6 | 0.0014 | 40 |
| Lagodon rhomboides CPH | 7/04 | $\leq$ 71.7 | 0.0008 | 93 | NS | NS | 43 |
| Lagodon rhomboides CPH | 9/04 | $\leq$ 87.4 | 0.0005 | 95 | NS | NS | 45 |

Appendix Q. Mean frequency of occurrence for Fundulus heteroclitus and Lagodon rhomboides at set positions from salt marsh during ebb tide eel pot collections. For each species comparisons between mainland salt marsh (mainland) and island salt marsh (island) salt marsh types that are significantly different from one another are indicated by asterisks ($p \leq 0.05$).

| | | Location | | | |
|---|---|---|---|---|---|
| | | 25 m | | 100 m | |
| Species | Date | Mainland | Island | Mainland | Island |
| Fundulus heteroclitus | 11/03 | 0.7 | 0.3 | 0.2 | 0.2 |
| Fundulus heteroclitus | 1/04 | 0.2 | 0.0 | 0.0 | 0.0 |
| Fundulus heteroclitus | 3/04 | 0.7 | 0.2 | 0.2 | 0.0 |
| Fundulus heteroclitus | 5/04 | 0.0 | 0.0 | 0.0 | 0.0 |
| Fundulus heteroclitus | 7/04 | 0.3 | 0.0 | 0.0 | 0.0 |
| Fundulus heteroclitus | 9/04 | 0.0 | 0.0 | 0.0 | 0.0 |
| Lagodon rhomboides | 11/03 | 0.8 | 0.5 | 0.7 | 0.8 |
| Lagodon rhomboides | 1/04 | 0.0 | 0.0 | 0.0 | 0.0 |
| Lagodon rhomboides | 3/04 | 0.0 | 0.0 | 0.0 | 0.0 |
| Lagodon rhomboides | 5/04 | 1.0* | 0.3* | 0.8 | 0.6 |
| Lagodon rhomboides | 7/04 | 1.0 | 0.8 | 1.0 | 1.0 |
| Lagodon rhomboides | 9/04 | 1.0 | 0.8 | 0.8 | 1.0 |

Appendix R. Rank of fish and decapod species observed in gill net collections from mainland, large island and small island salt marsh based on percent abundance and biomass.

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---------- Year One ----------

----------ABUNDANCE----------

| ----------Mainland---------- | | | ----------Large Island---------- | | | ----------Small Island---------- | | |
|---|---|---|---|---|---|---|---|---|
| RANK | SPECIES | % | RANK | SPECIES | % | RANK | SPECIES | % |
| 1 | Mugil cephalus | 24.2 | 1 | Paralichthys albigutta | 22.2 | 1 | Brevoortia tyrannus | 27.9 |
| 1 | Sciaenops ocellatus | 24.2 | 2 | Mugil cephalus | 18.5 | 2 | Mustelus canis | 25.6 |
| 2 | Lagodon rhomboides | 9.1 | 3 | Brevoortia tyrannus | 11.1 | 3 | Leiostomus xanthurus | 11.6 |
| 2 | Paralichthys dentatus | 9.1 | 3 | Leiostomus xanthurus | 11.1 | 4 | Mugil cephalus | 7.0 |
| 2 | Pomatomus saltatrix | 9.1 | 4 | Orthopristis chrysoptera | 7.4 | 4 | Paralichthys lethostigma | 7.0 |
| 3 | Brevoortia tyrannus | 3.0 | 4 | Paralichthys lethostigma | 7.4 | 5 | Elops saurus | 4.6 |
| 3 | Cynoscion regalis | 3.0 | 4 | Squalus acanthias | 7.4 | 5 | Paralichthys dentatus | 4.6 |
| 3 | Dasyatis americana | 3.0 | 5 | Dasyatis americana | 3.7 | 5 | Sciaenops ocellatus | 4.6 |
| 3 | Micropogon undulatus | 3.0 | 5 | Lagodon rhomboides | 3.7 | 6 | Lagodon rhomboides | 2.3 |
| 3 | Mustelus canis | 3.0 | 5 | Pomatomus saltatrix | 3.7 | 6 | Orthopristis chrysoptera | 2.3 |
| 3 | Opsanus tau | 3.0 | 5 | Strongylura notata | 3.7 | 6 | Pomatomus saltatrix | 2.3 |
| 3 | Paralichthys albigutta | 3.0 | | | | | | |
| 3 | Paralichthys lethostigma | 3.0 | | | | | | |

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Appendix R. (Continued).

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---------- Year One ----------

---------- BIOMASS----------

| ----------Mainland---------- | | | ----------Large Island---------- | | | ----------Small Island---------- | | |
|---|---|---|---|---|---|---|---|---|
| RANK | SPECIES | % | RANK | SPECIES | % | RANK | SPECIES | % |
| 1 | Sciaenops ocellatus | 68.4 | 1 | Squalus acanthias | 30.6 | 1 | Paralichthys lethostigma | 21.0 |
| 2 | Mugil cephalus | 14.0 | 2 | Pomatomus saltatrix | 27.5 | 2 | Mustelus canis | 18.2 |
| 3 | Pomatomus saltatrix | 4.5 | 3 | Paralichthys lethostigma | 18.0 | 3 | Sciaenops ocellatus | 17.9 |
| 4 | Cynoscion regalis | 2.9 | 4 | Mugil cephalus | 8.0 | 4 | Brevoortia tyrannus | 11.3 |
| 5 | Dasyatis americana | 2.0 | 5 | Dasyatis americana | 4.0 | 5 | Elops saurus | 10.3 |
| 5 | Lagodon rhomboides | 2.0 | 6 | Paralichthys albigutta | 3.1 | 6 | Paralichthys dentatus | 6.7 |
| 6 | Opsanus tau | 1.6 | 7 | Strongylura notata | 3.0 | 7 | Mugil cephalus | 6.3 |
| 7 | Mustelus canis | 1.3 | 8 | Leiostomus xanthurus | 2.5 | 8 | Leiostomus xanthurus | 4.3 |
| 8 | Paralichthys dentatus | 1.1 | 9 | Orthopristis chrysoptera | 1.4 | 9 | Pomatomus saltatrix | 1.8 |
| 9 | Brevoortia tyrannus | 0.7 | 10 | Brevoortia tyrannus | 1.1 | 10 | Orthopristis chrysoptera | 1.2 |
| 10 | Micropogon undulatus | 0.6 | 11 | Lagodon rhomboides | 0.6 | 11 | Lagodon rhomboides | 0.8 |
| 11 | Paralichthys lethostigma | 0.4 | | | | | | |
| 12 | Paralichthys albigutta | 0.3 | | | | | | |

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Appendix R. (Continued).

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---------- Year Two ----------

----------ABUNDANCE----------

| ----------Mainland---------- | | | ----------Large Island---------- | | | ----------Small Island---------- | | |
|---|---|---|---|---|---|---|---|---|
| RANK | SPECIES | % | RANK | SPECIES | % | RANK | SPECIES | % |
| 1 | Mugil cephalus | 25.0 | 1 | Brevoortia tyrannus | 53.2 | 1 | Mustelus canis | 68.8 |
| 2 | Brevoortia tyrannus | 17.5 | 2 | Peprilus triacanthus | 21.0 | 2 | Mugil cephalus | 10.4 |
| 2 | Paralichthys lethostigma | 17.5 | 3 | Mugil cephalus | 8.9 | 3 | Leiostomus xanthurus | 4.2 |
| 3 | Leiostomus xanthurus | 10.0 | 4 | Pomatomus saltatrix | 4.0 | 3 | Pogonias cromis | 4.2 |
| 4 | Anchoa hepsetus | 7.5 | 5 | Leiostomus xanthurus | 3.2 | 4 | Brevoortia tyrannus | 2.1 |
| 4 | Lagodon rhomboides | 7.5 | 5 | Mustelus canis | 3.2 | 4 | Lagodon rhomboides | 2.1 |
| 5 | Paralichthys dentatus | 5.0 | 6 | Lagodon rhomboides | 2.4 | 4 | Micropogon undulatus | 2.1 |
| 6 | Gymnura micrura | 2.5 | 7 | Orthopristis chrysoptera | 1.6 | 4 | Pomatomus saltatrix | 2.1 |
| 6 | Orthopristis chrysoptera | 2.5 | 8 | Paralichthys lethostigma | 0.8 | 4 | Sciaenops ocellatus | 2.1 |
| 6 | Paralichthys albigutta | 2.5 | 8 | Paralichthys dentatus | 0.8 | 4 | Sphyrna tiburo | 2.1 |
| 6 | Synodus foetens | 2.5 | 8 | Sciaenops ocellatus | 0.8 | | | |

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Appendix R. (Continued).

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---------- Year Two ----------

---------- BIOMASS----------

| ----------Mainland---------- | | | ----------Large Island---------- | | | ----------Small Island---------- | | |
|---|---|---|---|---|---|---|---|---|
| RANK | SPECIES | % | RANK | SPECIES | % | RANK | SPECIES | % |
| 1 | Gymnura micrura | 40.8 | 1 | Brevoortia tyrannus | 46.9 | 1 | Mustelus canis | 41.0 |
| 2 | Paralichthys lethostigma | 18.2 | 2 | Mugil cephalus | 13.5 | 2 | Sphyrna tiburo | 19.8 |
| 3 | Mugil cephalus | 17.5 | 3 | Sciaenops ocellatus | 10.4 | 3 | Pogonias cromis | 13.6 |
| 4 | Paralichthys albigutta | 6.8 | 4 | Peprilus triacanthus | 8.3 | 4 | Sciaenops ocellatus | 8.8 |
| 5 | Paralichthys dentatus | 6.3 | 5 | Pomatomus saltatrix | 5.4 | 5 | Mugil cephalus | 8.3 |
| 6 | Brevoortia tyrannus | 4.3 | 6 | Mustelus canis | 4.9 | 6 | Pomatomus saltatrix | 5.1 |
| 7 | Leiostomus xanthurus | 3.2 | 7 | Leiostomus xanthurus | 3.5 | 7 | Leiostomus xanthurus | 1.5 |
| 8 | Lagodon rhomboides | 1.7 | 8 | Paralichthys lethostigma | 2.8 | 8 | Micropogon undulatus | 0.8 |
| 9 | Orthopristis chrysoptera | 0.7 | 9 | Lagodon rhomboides | 2.1 | 9 | Brevoortia tyrannus | 0.7 |
| 10 | Synodus foetens | 0.2 | 10 | Orthopristis chrysoptera | 1.7 | 10 | Lagodon rhomboides | 0.2 |
| 11 | Anchoa hepsetus | 0.1 | 11 | Paralichthys dentatus | 0.4 | | | |

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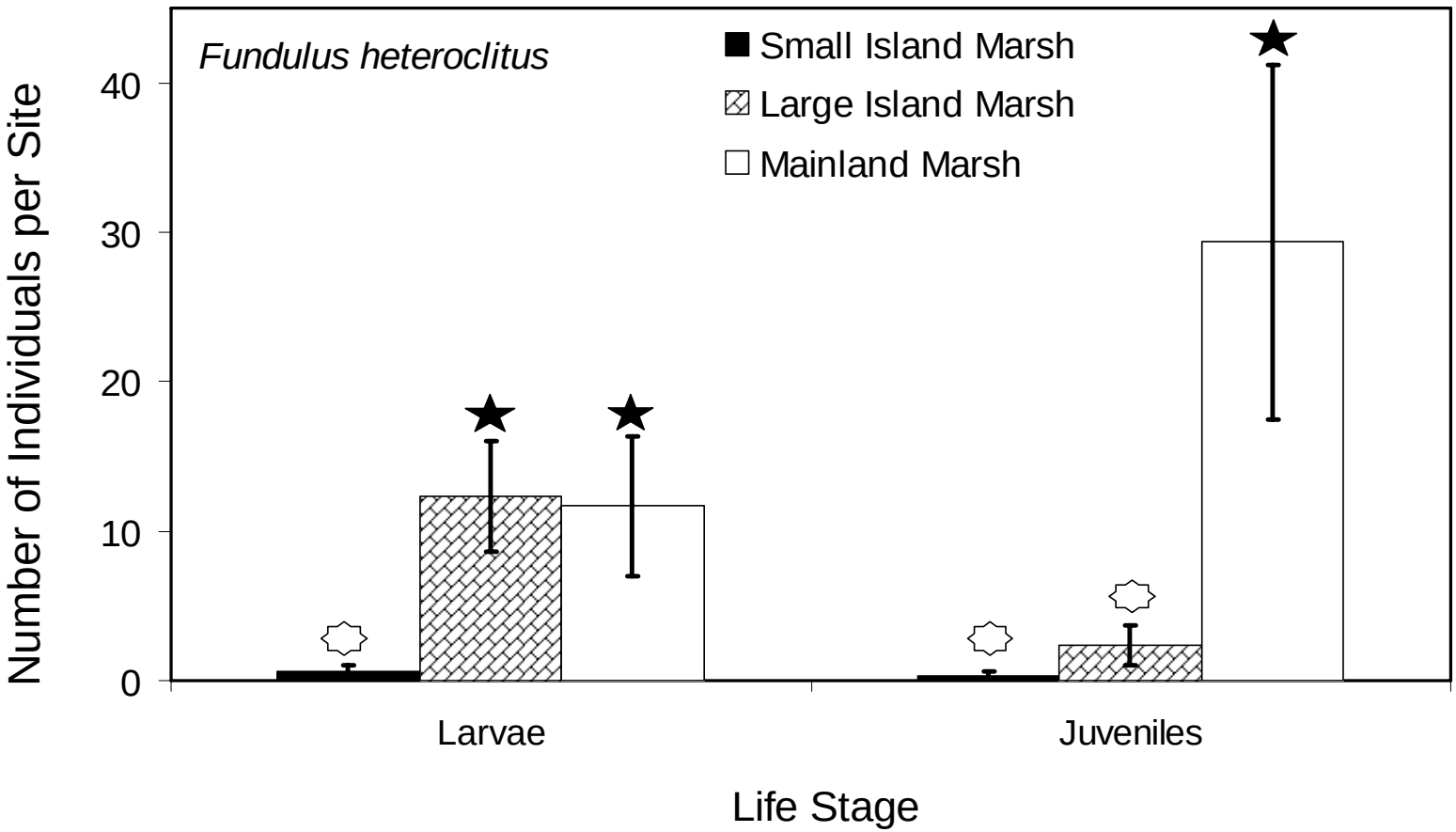
Appendix S. Mean number of Fundulus heteroclitus larvae and juveniles collected per salt marsh type during Breder and pit trap collections. For each life history stage, salt marsh mean comparisons among salt marsh types that are significantly different ($p \leq 0.05$) from one another are indicated by a different symbol type. One standard error is indicated by the error bars.


Fundulus heteroclitus
Small Island Marsh
Large Island Marsh
Mainland Marsh
Number of Individuals per Site
0
10
20
30
40
Larvae
Juveniles
Life Stage